5-2013

Improving Surrogate Monitoring Techniques for Suspended Sediment

Brant R. Whiting
Utah State University

UtahStateUniversity
MERRILL-CAZIER LIBRARY

IMPROVING SURROGATE MONITORING TECHNIQUES FOR

SUSPENDED SEDIMENT

by

Brant R. Whiting

A thesis submitted in partial fulfillment
of the requirements for the degree

of

MASTER OF SCIENCE

in

Civil and Environmental Engineering

Approved:

_____   _____
Dr. Jeffery S. Horsburgh   Dr. David K. Stevens
Major Professor   Committee Member

_____   _____
Dr. Patrick Belmont   Dr. Mark R. McLellan
Committee Member   Vice President for Research and
                                         Dean of the School of Graduate Studies

UTAH STATE UNIVERSITY
Logan, Utah

2013

ABSTRACT

Improving Surrogate Monitoring Techniques for Suspended Sediment

by

Brant R. Whiting, Master of Science

Utah State University, 2013

Major Professor: Dr. Jeffery S. Horsburgh
Department: Civil and Environmental Engineering

In the United States, suspended sediment is cited as the most common impairment to water quality. In addition to suspended sediment being a primary pollutant of concern, many other contaminants and nutrients are associated with suspended sediment particles. Suspended sediment has significant ecological impacts stemming from changes in physical, chemical and biological characteristics of surface waters and the benthic environment. The ability to accurately quantify suspended sediment concentrations and fluxes at appropriate temporal and spatial scales is critical in assessing whether streams are meeting their designated beneficial uses and in implementing and evaluating watershed management and mitigation plans and restoration efforts. Currently, new methods for quantifying suspended sediment concentrations and fluxes use least squares regression between turbidity and TSS (total suspended solids) and have been shown to be site-specific and are affected by several factors, including the size and characteristics of suspended sediment particles; and a single in situ sensor and grab samples of TSS may

not be representative of conditions throughout the entire stream cross section. In this research we used turbidity as a surrogate for TSS at six locations in the Little Bear River, Utah, U.S.A. We also examined the variability in discrete versus width and depth integrated TSS sampling at two sites to develop methods for accounting for these within the surrogate relationships and improve resulting estimates of TSS and fluxes. TSS concentration quantiles were used to assess—in probabilistic terms—the duration and magnitude of potential water quality criteria exceedance. Findings highlight that among some monitoring locations with wide spatial distribution, turbidity-TSS relationships are not site-specific for the more frequent (90th percentile) but lower (<50 NTU) turbidity values. Comparisons of area-weighted point measures of turbidity and width and depth integrated TSS samples revealed that suspended sediment flux is homogenous at their respective stream cross sections for 90% and 99% of the time at sites 2 and 6, respectively. The results are applicable in the determination of attainment or exceedance of a given water quality criterion.

(162 pages)

PUBLIC ABSTRACT

Improving Surrogate Monitoring Techniques for Suspended Sediment

Brant R. Whiting

The quality of water of our nation's rivers and streams is important to many vital

uses including drinking water treatment, recreation, and the natural environment. Water

quality can be severely impacted by the quantity and type of suspended sediment found

therein. Because suspended sediment can be associated with many other contaminants

that degrade water quality, it is noted as the most common impairment to water quality in

the United States. Suspended sediment can cause significant ecological impacts to the

chemical and biological characteristics of surface waters. The ability to accurately

quantify suspended sediment concentrations at the appropriate time(s) and location(s) is

critical in assessing whether streams are meeting their designated beneficial uses and in

implementing and evaluating watershed management and mitigation plans and restoration

efforts. Currently, new methods for quantifying suspended sediment concentrations use

mathematical and statistical techniques to relate turbidity and suspended sediment and

have been shown to be affected by several factors, including the size and characteristics

of suspended sediment particles. In this research we used turbidity as a surrogate

(substitute) for suspended sediment at six locations in the Little Bear River, Utah, U.S.A.

We also examined the differences between single-point and width and depth integrated

suspended sediment sampling at two sites. This was used to develop a method to

account for the differences and improve the resulting estimates of suspended sediment

concentrations. Statistical techniques were used to assess—in probabilistic terms—the duration and magnitude of potential water quality criteria exceedance. Findings highlight that among some monitoring locations with wide geographic distances, turbidity-suspended sediment relationships are not site-specific for the more frequent (90th percentile) but lower (<50 NTU) turbidity values. Comparisons of point measures of turbidity and width and depth integrated suspended sediment samples revealed that suspended sediment is homogenous at their respective stream cross sections for 90% and 99% of the time at sites 2 and 6, respectively. The results are applicable to water managers who are charged with the determination of attainment or exceedance of water quality standards.

Brant R. Whiting

ACKNOWLEDGMENTS

I would like to express special appreciation to Dr. Jeff Horsburgh for providing me the opportunity to carry out this research and for encouraging me to explore my interests. I very much appreciate his patience and mentorship through my graduate studies. I am also very appreciative to Dr. Horsburgh, the Utah Water Research Laboratory, and the USU College of Engineering for the generous funding that was made available for my graduate schooling and research.

I would like to thank my graduate committee members, Dr. David Stevens and Dr. Patrick Belmont, for the time given to review my work and provide meaningful feedback and guidance. I am grateful for the materials and laboratory equipment that they made available to me in order to carry out critical aspects of my research. I appreciate their professionalism, instruction, and encouragement to do my very best.

I am grateful to Charles and Cathy Whiting and Rhett and Alice James who have been instrumental in providing financial and moral encouragement during my extended university experience. Their prayers and kind words were inspiring and supportive.

I am indebted immensely to my wife, Brooke, for her enduring patience, love, and get-it-done encouragement. I thank my children, Hannah and Sam, for their smiles and hugs at bedtime when I usually answered "yes" to their nightly question: "Dad, do you have to go back to the water lab?" The love and prayers of my family, together with the blessings of heaven, made this work possible.

Brant R. Whiting

CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1

INTRODUCTION

As of 2009, only five states within the United States had numeric water quality criteria for suspended sediment in the nation's surface waters. Despite the lack of established numeric water quality criteria for suspended sediment in the United States, suspended sediment (including other solid-phase particulates) is cited as the most common impairment to water quality (U.S. EPA, 2008). Indeed, suspended sediment has significant ecological impacts stemming from changes in physical, chemical and biological characteristics of surface waters and the benthic environment (Rubin, 1995; Henley et al., 2000; Owens et al., 2001; Soulsby et al., 2001; Carter et al., 2006; Schoellhamer et al., 2007). Furthermore, suspended sediment is not only a primary pollutant of concern, but it can also be associated with many other contaminants and nutrients including phosphorus (Kronvang et al., 1997; Schoellhamer et al., 2007; Horowitz, 2008). The ability to accurately quantify suspended sediment concentrations and fluxes at appropriate temporal and spatial scales is critical to assessing whether streams are meeting their designated beneficial uses and in implementing and evaluating watershed management and mitigation plans.

Techniques for quantifying suspended sediment concentration have typically included traditional grab sampling methods, which lack sufficient temporal frequency to capture daily, weekly and monthly fluctuations in these concentrations and fluxes (Christensen et al., 2002; Tomlinson and De Carlo, 2003; Kirchner, 2004; Horsburgh et al., 2010; Jones et al., 2010). Currently, new methods for quantifying suspended sediment concentrations and fluxes are being used throughout the United States that

convert high frequency, in situ measurements of surrogates such as turbidity into high frequency estimates of suspended sediment concentration using statistical regression techniques that relate in situ turbidity measurements and much lower frequency grab samples of suspended sediment (Schoellhamer and Wright, 2003; Rasumussen et al., 2009; Jastram et al., 2010). One potential application of the large, continuous datasets that result is assessing whether water quality criteria are being exceeded. Doing so requires an examination of the uncertainty associated with the predicted suspended sediment concentrations.

Although the use of surrogate methods is becoming more widespread, there are some limitations to the approach. Turbidity-based suspended sediment concentration estimates are affected by two major factors (1) particle characteristics such as size, density, organic/mineral composition, and sediment and water color that affect the measured turbidity response and (2) point based sampling compared to width and depth-integrated sampling techniques (Jastram et al., 2010). Because turbidity is an optical measurement of the scattering of light as it passes through a given water sample, particle size, color and density have direct influence on turbidity and it's correlation with suspended sediment. Turbidity sensor output has been shown to vary by a factor of four for samples with identical suspended sediment concentrations but different particle characteristics (Gippel, 1995; Schoellhamer et al., 2003). It is hypothesized that this is one of the main reasons that relationships between turbidity and suspended sediment concentration are site specific. Additionally, in most applications surface-grab sampling methods and in situ turbidity measurements are made at a single sensor location in the

stream channel. While it is known that a single-point measurement of turbidity and suspended sediment may vary significantly from width and depth-integrated sampling methods (Martin et al., 1992; Horowitz, 2008; Jastram, 2010), it may be possible to account for these differences by developing a correction factor that can be used in regression model equations to account for this variability at a particular site.

This research sought to improve surrogate methods for estimating suspended sediment concentrations by: 1) investigating and suggesting ways to account for the impacts of sediment characteristics on turbidity measurements and subsequent concentrations estimated from turbidity; and 2) investigating the variability in concentrations between point and cross sectional sampling methods as well as methods for mitigating differences in estimation of concentrations and loads.

This research sought to identify the physical properties of suspended sediment that contribute to the site specific nature of relationships between turbidity and suspended sediment concentration and investigated their quantification and inclusion in regression models. This was done in effort to understand the effects of sediment particle characteristics on turbidity and improve the precision and accuracy of suspended sediment concentration and flux estimates derived from turbidity.

This research also investigated the appropriateness of using single-point samples of turbidity and suspended sediment concentration as representative of the entire stream cross section, as opposed to using depth and width-integrated methods. Examining the differences between the single-point measurements and the more integrative, cross sectional methods for measuring suspended sediment concentration enabled us to develop

a method for correcting estimates of concentration and load derived from a single turbidity sensor to be more representative of conditions across the entire channel.

Because of the existing continuous monitoring infrastructure and available water quality and environmental data, the Little Bear River watershed of northern Utah, USA, offered an ideal environmental observatory in which the proposed research could be tested over a range of spatial conditions and time scales. This research demonstrated improvements to the methodology for quantifying high frequency measures of suspended sediment concentrations using surrogate techniques by accounting for the effects of particle characteristics on measurements of turbidity. This research allowed us to examine differences between discrete and cross section integrated suspended sediment concentrations and to develop a correction factor to account for differences between the discrete-based regression computed suspended sediment concentrations and the cross section integrated concentrations. Finally, this research demonstrated how quantification of the uncertainty associated with high frequency estimates of suspended sediment concentration can be useful in the determination of attainment or exceedance of a water quality criterion. These results are of interest to both management and policy based practices and programs.

References

Carter, J., Walling, D. E., Owens, P. N., and Leeks, G. J. L. (2006). "Spatial and temporal variability in the concentration and speciation of metals in suspended sediment transported by the River Aire, Yorkshire, UK." *Hydrological Processes*, 20(14), 3007-3027.

Christensen, V., Rasmussen, P., and Ziegler, A. (2002). "Real-time water quality monitoring and regression analysis to estimate nutrient and bacteria concentrations in Kansas streams." *Water Science and Technology*, <http://ks.water.usgs.gov/pubs/reports/vgc.0610.html> (July 15, 2011).

Gippel, C. J. (1995). "Potential of turbidity monitoring for measuring the transport of suspended solids in streams." *Hydrological Processes*, 9(1), 83-97.

Henley, W. F., Patterson, M. A., Neves, R. J., and Lemly, A. D. (2000). "Effects of sedimentation and turbidity on lotic food webs: A concise review for natural resource managers." *Reviews in Fisheries Science*, (August 2011), 37-41.

Horowitz, A. J. (2008). "Determining annual suspended sediment and sediment-associated trace element and nutrient fluxes." *The Science of the Total Environment*, Elsevier B.V., 400(1-3), 315-43.

Horsburgh, J. S., Spackman Jones, A., Stevens, D. K., Tarboton, D. G., and Mesner, N. O. (2010). "A sensor network for high frequency estimation of water quality constituent fluxes using surrogates." *Environmental Modeling & Software*, Elsevier Ltd, 25(9), 1031-1044.

Jastram, J. D., Zipper, C. E., Zelazny, L. W., and Hyer, K. E. (2010). "Increasing precision of turbidity-based suspended sediment concentration and load Estimates." *Journal of Environmental Quality*, 1306.

Jones, A. S., Stevens, D. K., Horsburgh, J. S., and Mesner, N. O. (2010). "Surrogate Measures for providing high frequency estimates of total suspended solids and total phosphorus concentrations." *Journal of the American Water Resources Association*, 47(2), 239-253.

Kirchner, J. W., Feng, X., Neal, C., and Robson, A. J. (2004). "The fine structure of water-quality dynamics: The (high-frequency) wave of the future." *Hydrological Processes*, 18(7), 1353-1359.

Kronvang, B., Laubel, A., and Grant, R. (1997). "Suspended sediment and particulate phosphorus transport and delivery pathways in an arable catchment, Gelbæk Stream, Denmark." *Hydrological Processes*, 11(6), 627-642.

Martin, G., Smoot, J., and White, K. D. (1992). "A comparison of surface-grab and cross sectionally integrated stream-water-quality sampling methods." *Water Environment Research*, 64(7), 866-876.

Owens, P. N., Walling, D. E., Carton, J., Meharg, A. A., Wright, J., and Leeks, G. J. L. (2001). "Downstream changes in the transport and storage of sediment-associated contaminants (P, Cr and PCBs) in agricultural and industrialized drainage basins." *The Science of the Total Environment*, 266(1-3), 177-86.

Rasmussen, P. P., Gray, J. R., Glysson, G. D., and Ziegler, A. C. (2009). "Guidelines and procedures for computing time-series suspended-sediment concentrations and loads from in-stream turbidity-sensor and streamflow data." U.S. Geological Survey Techniques and Methods book 3, chap. C4, 53 p.

Rubin, J. F. (1995). "Estimating the success of natural spawning of salmonids in streams." *Journal of Fish Biology*, Wiley Online Library, 46(4), 603-622.

Schoellhamer, D.H., and Wright, S. A. (2003). "Continuous measurement of suspended-sediment discharge in rivers by use of optical backscatterance sensors." *IAHS Publication*, (June 2002), 28-36.

Schoellhamer, D. H., Mumley, T. E., and Leatherbarrow, J. E. (2007). "Suspended sediment and sediment-associated contaminants in San Francisco Bay." *Environmental Research*, 105(1), 119-31.

Soulsby, C., Youngson, A. F., Moir, H. J., and Malcolm, I. A. (2001). "Fine sediment influence on salmonid spawning habitat in a lowland agricultural stream: A preliminary assessment." *The Science of the Total Environment*, 265(1-3), 295-307.

Tomlinson, M. S., and De Carlo, E. H. (2003). "The need for high resolution time series data to characterize Hawaiian streams." *Journal of the American Water Resources Association*, 39(1), 113-123.

U.S. Environmental Protection Agency (U.S. EPA). (2008). "National summary of impaired waters and TMDL information." <http://iaspub.epa.gov/waters10/attains_nation_cy.control?p_report_type=T#causes_303d> (Nov. 1, 2011).

CHAPTER 2

LITERATURE REVIEW

2.1 Study Site Description

The Little Bear River watershed is located in the south end of Cache Valley, Utah

and encompasses roughly 740 km^2. The headwaters form in the Bear River Mountain

Range, and the river drains to Cutler Reservoir on the west side of Cache Valley.

Elevations within the watershed range from 2865 m to 1345 m. Land use within the

upper watershed is comprised of mostly forest and grazing, while the lower portion of the

watershed is made up primarily of agriculture with irrigated crop lands and some urban

development. Hydrology of the watershed is dominated by spring snowmelt, with water

in the upper watershed being stored within two reservoirs, Porcupine Reservoir in the

upper East Fork and Hyrum Reservoir near Paradise, Utah (Figure 2.1), before being

released and diverted into agricultural irrigation canals for use during the irrigation

season.

Since 2005, research conducted by the Utah Water Research Laboratory at Utah

State University has facilitated the installation of seven continuous water quality

monitoring stations (Figure 2.1). Table 2.1 summarizes the locations and descriptions of

the seven water quality monitoring sites located in the watershed along with the dates

from which the respective measured parameters are available.

Five of these stations (sites 1-5) are located above Hyrum reservoir where land

use is dominated by open range grazing with some urban and agriculture use and stream

channels have higher slopes and velocities. The lower portion of the watershed includes

two monitoring sites (one below Hyrum Dam and one at the terminus of the watershed near the confluence with Cutler Reservoir) where the land use near the river is dominated by agriculture. Additionally, within this lower section of the river soils are predominantly fine grained lacustrine sediments, and one regulated point source (Wellsville municipal wastewater treatment facility) contributes to the river. The lower portion of the river also exhibits lower channel slopes with meandering planform and slower flows.

The Little Bear watershed is interesting for research purposes because it has been affected by many anthropogenic influences such as a wide mix of land uses (forest, grazing, agriculture, urban), hydrologic modifications (reservoir dam releases, diversions canals), nonpoint sources (agriculture, animal feed lots) and point sources (Wellsville WWTP) that have caused elevated levels of sediment and nutrients. The Little Bear was listed on Utah's 303(d) list for impaired water bodies due to phosphorus and stream channel alteration. Many improvements have been completed in the Little Bear watershed as a result of a Total Maximum Daily Load (TMDL) implementation starting in 2000 (Utah DEQ, 2006). Because of the continuous water quality monitoring infrastructure in the Little Bear, the combined impact of these anthropogenic influences, varying climate conditions and watershed behavior can be examined.

2.2 Existing Data Collection Infrastructure

The Little Bear River monitoring system was established as one of ten test bed sites as part of the WATERS network (http://www.watersnet.org) project funded by the National Science Foundation. The Little Bear River was instrumented at seven locations

between the years of 2005 and 2007 with water quality monitoring equipment operating continuously. See Figure 2.1 for map for location of sites and Table 2.1 for summary of water quality monitoring sensors.

Water quality and stream flow data are collected on the Little Bear River using the sensors that are summarized in Table 2.2. A Hydrolab MS5 water quality sonde, Forest Technologies DTS 12 turbidity sensor, and ACData Solutions SPXD-600 pressure transducer. The Hydrolab sondes record measurements of water temperature, dissolved oxygen, specific conductance and pH; the turbidity sensor measures turbidity values and water temperature; and the pressure transducer records stage. Campbell Scientific dataloggers (e.g., CR200 and CR800) are programmed to initiate measurements every five minutes (turbidity sensor every 30 minutes). These measurements are averaged over 30 minutes and reported as 30 minute average values.

Meteorological monitoring is continuously conducted at four weather stations in the Little Bear watershed. Each station is equipped with a CS 215 temperature and relative humidity probe, CS100 barometric pressure sensor, TE525 tipping bucket rain gage, CS300 pyranometer, 03001 R.M. Young Wind Sentry set, CS SR50A Sonic Ranging Sensor and soil moisture probes. See Table 2.3 for a summary of instrumentation at the weather stations.

Continuous data from each of the seven water quality monitoring sites and four weather stations are transmitted daily via radio telemetry and TCP/IP networks to the Utah Water Research Laboratory where they are uploaded to a central database. This data can be queried and downloaded through http://littlebearriver.usu.edu.

Data from the Paradise and Mendon sites have been used to develop high frequency estimates of total suspended solids (TSS) and total phosphorus (TP) concentrations using turbidity as a surrogate (Jones et al., 2010). Preliminary work has also been done to develop estimates of TSS concentrations and related loadings from turbidity data at the remaining five sites (Horsburgh et al., 2010). Differences in relationships developed at each site were observed in this preliminary work, but the reasons for these differences were not investigated in depth. The continuance of data collection over the past several years has contributed to a rich database of high resolution water quality and environmental data for the Little Bear watershed. Because of the existing infrastructure and available data, the Little Bear River is an ideal environmental observatory in which the proposed research may be tested over a wide range of spatial conditions and time scales.

2.3 Fluvial Suspended Sediment

In 2008 the U.S Environmental Protection Agency (U.S. EPA) reported that sediments (and other solid-phase particulate) are the most common pollutant to U.S surface waters (U.S. EPA, 2008). This is also true in Utah, where 1,517 out of 2,975 stream miles (51 %) listed as impaired on the State's 303(d) list are listed for sediment related impairments (Utah DEQ, 2006). Indeed, sediment-related problems in North America are causing physical, chemical and biological degradation to water quality and are estimated to cost $20 - $50 billion each year (Pimentel et al., 1995; Osterkamp et al., 1998). Problems associated with sediment include potential impacts to: drinking water treatment processes (AWWA, 1990), fate and transport of heavy metals (Carter et al.,

2006; Schoellhamer et al., 2007), recreational uses, reservoir storage and hydraulic

operations (Pruitt, 2003), barge transportation and navigation lanes (U.S. Army Corps of

Engineers, 2011), light suppression effects on stream bed vegetation, ecological function

of aquatic habitat (Henley et al., 2000), food webs, and fish spawning beds (Rubin, 1995;

Soulsby et al., 2001). Sediments can also determine the transport and fate of pollutants as

they are carried through the river system and are deposited and resuspended over time

(Owens et al., 2001).

Fluvial suspended sediment is the solid particulate matter suspended in the water

column and ranging in size from 1μm – 1 mm (Chapra, 1997), though the actual range

observed at any site is a subset of this range. Suspended sediment is quantified on a mass

(dry weight) per volume basis commonly in milligrams per liter (mg/L). Various

gravimetric methods exist for the determination of suspended sediment concentration in

surface waters and are usually dependent on the specific laboratory performing the

analysis (U. S. EPA, 1993; APHA, 1995; ASTM, 2007). Based on the method used, the

concentration of suspended sediment is reported as total suspended solids (TSS) or

suspended sediment concentration (SSC) in mg/L (Gray et al., 2000). In this proposal,

"suspended sediment concentration" is used to denote the actual concentration of fluvial

suspended sediment in the stream, whereas "SSC" is used to denote the suspended

sediment concentration in a sample as measured using the group of laboratory methods

that report "SSC." Gray et al., (2000) and Schoellhamer et al., (2007) report that SSC

and TSS measurements of natural waters are not always comparable and that the TSS

method is unreliable as an estimate of the concentration of fluvial suspended sediment in

natural waters when the sand-size ($> 63\mu m$) portion of the sample is greater than 25%. Preliminary analysis has shown that the majority of fluvial sediment in the Little Bear River consists of clays and silts ($< 63\mu m$) (Rasmussen et al., 2009; Jones et al., 2010), and therefore measurements of SSC and TSS are expected to be similar.

In most cases, the quantification of suspended sediment concentration involves field data collection methods that have been historically limited to traditional grab sampling techniques that are too infrequent to capture the temporal variation in suspended sediment concentrations and fluxes (Gippel, 1995; Christensen et al., 2002; Kirchner et al., 2004; Horsburgh et al., 2010). Suspended sediment loads are usually underestimated because they often miss storm events or other significant events during which much of the suspended load is transported (Tomlinson and De Carlo, 2003).

2.4 Turbidity as a Surrogate for Suspended Sediment Concentration

According to Ziegler (2002) turbidity is an optical measurement of the transparency of a solution due to the scattering, reflecting and attenuation of light by the suspended particulates and some dissolved matter. Higher turbidity values correspond to more intense scattering or attenuation. Turbidity has been well established as a surrogate measure of fluvial suspended sediment concentration in rivers throughout the world and is the most common surrogate used for measuring suspended sediment concentration in the United States (Gippel, 1995; Ziegler, 2002; Gray and Glysson, 2003; Pfannkuche and Schmidt, 2003; Carter et al., 2006; Stubblefield et al., 2007; Minella et al., 2008; Jastram et al., 2010; Jones et al., 2010). Turbidity-based TSS concentration estimation models have been developed in the Little Bear watershed. Jones et al., (2010) found significant

correlations (R^2 = 0.85 and 0.95) between turbidity and TSS at sites in the lower and upper portions of the watershed where turbidity was the only significant explanatory variable.

Advances in technology are decreasing the cost of turbidity sensors while improving field performance capabilities. Improvements in sensor technology allow for continuous in situ monitoring with fewer site visits for calibration and cleaning and lower instrument drift over time. Gray and Gartner (2009) reported that water quality monitoring systems that use turbidity as a surrogate for suspended sediment concentration not only provide a higher resolution of the timing and magnitude of concentrations and fluxes, but also have the potential to decrease costs and increase the safety of field data collection methods while increasing accuracy. Compared to other high frequency surrogate technologies such as laser diffraction or multifrequency acoustic backscatter units, Gray and Gartner (2009) stated that turbidity instruments can be as little as one sixth the cost of other technologies.

Furthermore, the viability of turbidity as a surrogate is supported by the interest shown by state agencies charged with monitoring water quality. The U.S. EPA (2002) reports that 29 states have established numeric criteria for turbidity and Gray and Glysson (2003) report that 32 states use turbidity or water clarity as a metric in the evaluation of water quality.

The characteristics of sediment particles, including size, shape, and composition as well as water color can affect turbidity measurements (Gippel, 1989; Foster et al., 1992; Davies-Colley and Smith, 2001) and their corresponding relationship to suspended

sediment concentrations (Jastram et al., 2010). The most significant factors in turbidity sensor response are particle-size distribution, organic matter content, specific surface area and particle density (Gippel, 1995; Davies-Colley and Smith, 2001; Jastram et al., 2010).

Because turbidity is a measure of light scatter, particle size has direct impact on sensor response. Smaller particles scatter light more efficiently because they have more reflective surface per unit of mass (Schoellhamer and Wright, 2003). Gippel (1995) found that particle size variations among water samples caused turbidity values to differ by a factor of four where the suspended sediment concentration was the same. Clifford et al., (1995) found significant variation in sensor response due to varying particle size distributions while testing an infrared turbidity sensor in laboratory and field settings. Gippel (1989) found suspended sediment loads that were uniformly composed of fine clays and organic colloids reduced variability in sensor response.

Properties of suspended sediment particles from organic sources (algae, plankton, micro-organisms) such as size and density cause light to scatter differently than mineral particles (Minella et al., 2008; Jastram et al., 2010). The portion of the suspended sediment that is made up of organic content is thought to contribute to variance in turbidity-suspended sediment concentration relationships (Schoellhamer and Wright, 2003; Pfannkuche and Schmidt, 2003; Minella et al., 2008). Jastram et al., (2010) found that the organic carbon content of SSC from the Roanoke River in southwest Virginia could be used as an additional explanatory variable in turbidity-based SSC model to improve the precision of SSC estimates.

Water color and colored particles also contribute to variance in turbidity response by absorbing light beams of the visual range, thus reducing the amount of scattered light detected by a turbidity sensor (Gippel, 1989; Anderson, 2005). Newer turbidity sensor models that use infrared light now mitigate the effects of color (Gippel, 1989; Clifford et al., 1995).

Additional sources of variation in turbidity-based estimates of suspended sediment concentrations are the sampling procedures that measure turbidity or collect grab samples from a single location in the stream channel (Martin et al., 1992; Edwards and Glysson, 1999; Horowitz, 2008; Jastram et al., 2010). Martin et al., (1992) found that single-point surface grab samples of SSC at four sites on the Kentucky River underrepresented width and depth-integrated samples over a two-year period using fifteen samples per site. For particles larger than 62 μm, single-point samples underestimated suspended sediment concentration by 51%. For particles smaller than 62 μm single-point samples underestimated suspended sediment concentration by 17%. These variations were attributed to differences in velocity across the stream channel, and incomplete mixing of groundwater, point sources, and upstream tributaries. This is, however, dependent on individual site conditions, as Jastram et al., (2010) demonstrated that differences in point and cross section-averaged measurements on the Roanoke River in Virginia were not statistically different.

2.5 New Methods for Measuring Fluvial Suspended Sediment

Use of acoustic doppler current profiler (ADCP) instruments has seen a steady rise over the past two decades as their use for measuring stream velocities and discharges

has expanded to derive suspended sediment concentrations via interpretation and conversion of acoustic backscatter/echo intensities (Reichel and Nachtnebel, 1994; Thorne and Hanes, 2001; Gray et al., 2001). Wall et al., (2006) found a high correlation (R^2 >0.85) between measured SSC and SSC estimated using ADCP measurements. They developed an empirical model that related SSC measured by point-integrating isokinetic samplers to measurements made using a boat-mounted ADCP on the Hudson River in New York. Byrne and Patino (2001) used data from streams in south Florida to generate a time series of SSC using acoustic backscatter as a surrogate. Acoustic backscatter was related to SSC at two sites using regression analysis and relationships were significant, with R^2 =0.91 for a fresh water site and R^2=0.87 in a stream with tidal influence (Gray et al., 2001).

2.6 Uncertainty in Fluvial Suspended Sediment Concentrations
   ## and Fluxes Derived from Continuous Data Sets

Suspended sediment concentrations can be quantified using turbidity measurements and linear regression analysis (Gippel, 1995; Minella et al., 2008 Rasmussen et al., 2009; Jastram et al., 2010). A resulting regression model provides a framework for which the quantification of uncertainty in continuous estimates of suspended sediment concentrations derived from turbidity can be determined. This generally involves developing prediction intervals that specify a range of values for which a future concentration value can be expected to fall within. Rasumussen et al., (2009) developed prediction intervals for estimates of turbidity-based SSC using 68 concurrent measurements of turbidity and SSC on the Little Arkansas River near

Sedgwick, Kansas. They calculated (with 90-percent confidence) the percentage of SSC values that exceeded a given water quality criterion during a period of about six years. The 90-percent prediction interval for their dataset was ± 41 percent of the computed SSC value.

2.7 Proposed Research

The overall purpose of this work is to improve the use of surrogate monitoring techniques in estimating suspended sediment concentrations and fluxes in order to better understand watershed response and behavior. Based on the above literature review, this work will first focus on assessing the generality of using turbidity as a surrogate for suspended sediment concentration and identifying potential confounding factors by investigating sediment particle characteristics that influence turbidity response. Second, it examines the uncertainty in estimates of continuous concentrations derived from turbidity and the potential implications for using the estimates to assess compliance with water quality criteria. Finally, this research looks into assessing whether high frequency estimates of concentration derived from turbidity, along with high frequency measurements of discharge and other water quality variables, can increase understanding of watershed function and assist in the development of monitoring and conservation practices aimed at managing sediment pollution.

References

American Public Health Association (APHA). (1995). "Standard methods for the examination of water and wastewater." American Public Health Association, American Water Works Association, and Water Pollution Control Federation, Washington, DC.

American Society for Testing and Materials (ASTM). (2007). "Standard test methods for determining sediment concentration in water samples." *ASTM International*, West Conshohcken, PA. D3977-97." 11.02 (Water (II)), 395-400.

American Water Works Association (AWWA). (1990). *Water quality and treatment: A handbook of community water supplies*. McGraw-Hill, New York.

Anderson, C. W. (2005). "Turbidity 6.7." U.S. Geological Survey Techniques of Water-Resources Investigations, book 9, chap. A6., sec. 6.7. <http://pubs.water.usgs.gov/twri9A6/> (October 2, 2011).

Byrne, M. J., and Patino, E. (2001). "Feasibility of using acoustic and optical backscatter instruments for estimating total suspended solids concentrations in estuarine environments." Proceedings of the Seventh Federal Interagency Sedimentation Conference; Reno, NV, March 25-29, 1(3), 135-138.

Carter, J., Walling, D. E., Owens, P. N., and Leeks, G. J. L. (2006). "Spatial and temporal variability in the concentration and speciation of metals in suspended sediment transported by the River Aire, Yorkshire, UK." *Hydrological Processes*, 20(14), 3007-3027.

Chapra, S. C. (1997). *Surface water-quality modeling*. McGraw-Hill, New York.

Christensen, V., Rasmussen, P., and Ziegler, A. (2002). "Real-time water quality monitoring and regression analysis to estimate nutrient and bacteria concentrations in Kansas streams." *Water Science and Technology*, <http://ks.water.usgs.gov/pubs/reports/vgc.0610.html> (July 15, 2011).

Clifford, N. J., Richards, K. S., Brown, R. A., and Lane, S. N. (1995). "Laboratory and field assessment of an infrared turbidity probe and its response to particle size and variation in suspended sediment concentration." *Hydrological Sciences Journal*, 40(6), 771-791.

Davies-Colley, R., and Smith, D. G. (2001). "Turbidity, suspended sediment, and water clarity: A Review." *Journal of the American Water Resources Association*, Wiley Online Library, 37(5), 1085-1101.

Edwards, T. K., and Glysson, G. D. (1999). "Field methods for measurement of fluvial sediment, U.S. Geological Survey Techniques of Water-Resources Investigations, book 3, chap. c2." <http://pubs.water.usgs.gov/twri3c2> (October 4, 2011).

Foster, I. D. L., Millington, R., and Grew, R. (1992). "The impact of particle size controls on stream turbidity measurement; some implications for suspended sediment yield estimation." Erosion and Sediment Transport Monitoring Programmes in River Basins, *IAHS Publication*, 210(210), 51-62.

Gippel, C. J. (1989). "The use of turbidimeters in suspended sediment research." *Hydrobiologia*, 176-177(1), 465-480.

Gippel, C. J. (1995). "Potential of turbidity monitoring for measuring the transport of suspended solids in streams." *Hydrological Processes*, 9(1), 83-97.

Gray, J. R., and Gartner, J. W. (2009). "Technological advances in suspended-sediment surrogate monitoring." *Water Resources Research*, (November 2008).

Gray, J. R., and Glysson, G. D. (2003). "Federal Interagency Workshop on Turbidity and other Sediment Surrogates." Proceedings of the Federal Interagency Workshop on Turbidity and Other Sediment Surrogates; U.S Geological Survey, Reno, NV, April 30-May 2, 2002.

Gray, J. R., Glysson, G. D., Turcios, L. M., and Schwarz, G. E. (2000). "Comparability of suspended sediment concentrations data and total suspended solids data." *U.S Geological Survey*, WRI 419,1 Reston, VA.

Gray, J. R., Melis, T. S., Patiño, E., Larsen, C., Topping, D. J., and Rasmussen, P. P. (2001). "U.S. Geological Survey Research on surrogate measurements for suspended sediment." Reston, VA, 95-100.

Henley, W. F., Patterson, M. A., Neves, R. J., and Lemly, A. D. (2000). "Effects of sedimentation and turbidity on lotic food webs: A concise review for natural resource managers." *Reviews in Fisheries Science*, (August 2011), 37-41.

Horowitz, A. J. (2008). "Determining annual suspended sediment and sediment-associated trace element and nutrient fluxes." *The Science of the Total Environment*, Elsevier B.V., 400(1-3), 315-43.

Horsburgh, J. S., Spackman Jones, A., Stevens, D. K., Tarboton, D. G., and Mesner, N. O. (2010). "A sensor network for high frequency estimation of water quality constituent fluxes using surrogates." *Environmental Modeling & Software*, Elsevier Ltd, 25(9), 1031-1044.

Jastram, J. D., Zipper, C. E., Zelazny, L. W., and Hyer, K. E. (2010). "Increasing precision of turbidity-based suspended sediment concentration and load Estimates." *Journal of Environmental Quality*, 1306.

Jones, A. S., Stevens, D. K., Horsburgh, J. S., and Mesner, N. O. (2010). "Surrogate Measures for providing high frequency estimates of total suspended solids and total phosphorus concentrations." *Journal of the American Water Resources Association*, 47(2), 239-253.

Kirchner, J. W., Feng, X., Neal, C., and Robson, A. J. (2004). "The fine structure of water-quality dynamics: the(high-frequency) wave of the future." *Hydrological Processes*, 18(7), 1353-1359.

Martin, G., Smoot, J., and White, K. D. (1992). "A comparison of surface-grab and cross sectionally integrated stream-water-quality sampling methods." *Water Environment Research*, 64(7), 866-876.

Minella, J. P. G., Merten, G. H., Reichert, M., and Clarke, R. T. (2008). "Estimating suspended sediment concentrations from turbidity measurements and the calibration problem." *Hydrological Processes*, 1830(August 2007), 1819-1830.

Osterkamp, W. R., Heilman, P., and Lane, L. J. (1998). "Economic considerations of a continental sediment-monitoring program." *International Journal of Sediment Research*, 13(4), 12-24.

Owens, P. N., Walling, D. E., Carton, J., Meharg, A. A., Wright, J., and Leeks, G. J. L. (2001). "Downstream changes in the transport and storage of sediment-associated contaminants (P, Cr and PCBs) in agricultural and industrialized drainage basins." *The Science of the Total Environment*, 266(1-3), 177-86.

Pfannkuche, J., and Schmidt, A. (2003). "Determination of suspended particulate matter concentration from turbidity measurements: Particle size effects and calibration procedures." *Hydrological Processes*, 17(10), 1951-1963.

Pimentel, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Shpritz, L., Fitton, L., Saffouri, R., and Blair, R. (1995). "Environmental and economic costs of soil erosion and conservation benefits." *Science* (New York, N.Y.), 267(5201), 1117-23.

Pruitt, B. A. (2003). "Uses of turbidity by states and tribes." Proceedings of the Federal Interagency Workshop on Turbidity and Other Sediment Surrogates, U.S Geological Survey, Reno, NV, April 30-May 2, 2002.

Rasmussen, P. P., Gray, J. R., Glysson, G. D., and Ziegler, A. C. (2009). "Guidelines and procedures for computing time-series suspended-sediment concentrations and loads from in-stream turbidity-sensor and streamflow data." U.S. Geological Survey Techniques and Methods book 3, chap. C4, 53 p.

Reichel, G., and Nachtnebel, H. P. (1994). "Suspended sediment monitoring in a fluvial environment: Advantages and limitations applying an Acoustic Doppler Current Profiler." *Water Research*, 28(4), 751-761.

Rubin, J. F. (1995). "Estimating the success of natural spawning of salmonids in streams." *Journal of Fish Biology*, Wiley Online Library, 46(4), 603-622.

Schoellhamer, D.H., and Wright, S. A. (2003). "Continuous measurement of suspended-sediment discharge in rivers by use of optical backscatterance sensors." *IAHS Publication*, (June 2002), 28-36.

Schoellhamer, D. H., Mumley, T. E., and Leatherbarrow, J. E. (2007). "Suspended sediment and sediment-associated contaminants in San Francisco Bay." *Environmental Research*, 105(1), 119-31.

Soulsby, C., Youngson, A. F., Moir, H. J., and Malcolm, I. A. (2001). "Fine sediment influence on salmonid spawning habitat in a lowland agricultural stream: A preliminary assessment." *The Science of the Total Environment*, 265(1-3), 295-307.

Stubblefield, A. P., Reuter, J. E., Dahlgren, R. A., and Goldman, C. R. (2007). "Use of turbidometry to characterize suspended sediment and phosphorus fluxes in the Lake Tahoe basin, California, USA." *Hydrological Processes*, Wiley Online Library, 21(3), 281-291.

Thorne, P., and Hanes, D. M. (2002). "A review of acoustic measurement of small-scale sediment processes." *Continental Shelf Research*, 22(4), 603-632.

Tomlinson, M. S., and De Carlo, E. H. (2003). "The need for high resolution time series data to characterize Hawaiian streams." *Journal of the American Water Resources Association*, 39(1), 113-123.

U.S. Army Corps of Engineers. (2011). "Maintenance dredging and the environment." <http://www.nab.usace.army.mil/whatwedo/dredging.htm> (Aug. 1, 2011).

U.S. Environmental Protection Agency (USEPA). (1993). "EPA Method 340.2." EPA, Washington, DC.

U.S. Environmental Protection Agency (U.S. EPA). (2002). "National water quality inventory 2000 Report." Washington, DC.

U.S. Environmental Protection Agency (U.S. EPA). (2008). "National summary of impaired waters and TMDL information." <http://iaspub.epa.gov/waters10/attains_nation_cy.control?p_report_type=T#causes_303d> (Nov. 1, 2011).

Utah Department of Environmental Quality (Utah DEQ). (2006). "Utah 2006 integrated report." Volume I - 305(b) Assessment.

Wall, G. R., Nystrom, E. A., and Litten, S. (2006). "Use of an ADCP to compute suspended sediment discharge in the tidal Hudson River, New York." U.S. Geological Survey Scientific Investigations Report 2006-5055, 16 p.

Ziegler, A. C. (2002). "Issues related to use of turbidity measurements as a surrogate for suspended sediment." Proceedings of the Federal Interagency Workshop on Turbidity and Other Sediment Surrogates, U.S Geological Survey, Reno, NV, April 30-May 2, 2002.

Table 2.1. Summary of continuous water quality monitoring sites in the LBR watershed

Site Number	Site Name	Date Range	Latitude	Longitude	Site Description
1	Upper South Fork	Oct. 2007-Present	41.4954	-111.818	Located in the upper portion of the watershed where land use is primarily forest and grazing
2	Lower South Fork	July 2007-Present	41.5065	-111.8151	Located below the confluence of the Upper South Fork and a major tributary, Davenport creek
3	East Fork	August 2007-Present	41.5292	-111.7993	Located below Porcupine reservoir, during the summer flows are predominantly from groundwater as the entire river is diverted to irrigation canals below this point
4	Avon/ Confluence	Nov. 2007-Present	41.5361	-111.8305	Located below the confluence of the East fork, Lower South fork sites and McMurdy Hollow
5	Paradise	June 2005-Present	41.5756	-111.8552	Located a few miles upstream of Hyrum reservoir
6	Wellsville	Nov. 2007-Present	41.6435	-111.9176	Located a few miles below Hyrum Reservoir, flows are controlled by dam releases and summer flows are dominated by groundwater flow
7	Mendon	August 2005-Present	41.7185	-111.9464	Located near the boundary of the watershed at Cutler reservoir. Water quality is affected by agriculture return flows and the Wellsville WWTP

Table 2.2. Summary of water quality sensor specifications

Variable	Sensor	Specifications
Turbidity	DTS-12 turbidity sensor Forest Technology Systems, Inc.	Accuracy: ± 2% 0-500 NTU and ± 4% 501-1600 NTU
Stage	SPXD-600 Pressure Transducer KWK Techologies, Inc.	Accuracy: ± 1% of the full measurement span
Water Temperature	Hydrolab MiniSonde5 Thermistor Hach Environmental, Inc.	Accuracy: ± 0.1 ^{o}C, resolution: 0.01 ^{o}C
pH	Hydrolab MiniSonde5 reference electrode Hach Environmental, Inc.	Accuracy: ± 0.2 pH units, resolution: 0.01 pH units
Dissolved oxygen concentration	Hydrolab MiniSonde5 optical LDO sensor Hach Environmental, Inc.	Accuracy: ± 0.1 mg L^{-1} at < 8mg $^{L-1}$ and ± 0.2mg $^{L-1}$ at > 8mg $^{L-1}$, resolution: 0.01 mg $^{L-1}$
Specific conductance	Hydrolab MiniSonde5 4-electrode, temperature compensated conductivity sensor Hach Environmental, Inc.	Accuracy: ± 0.5%, resolution: 0.001 mS cm^{-1}

Table 2.3. Summary of meteorological instrumentation specifications

Variable	Sensor	Specifications
Air Temperature	CS215 temperature and relative humidity sensor Campbell Scientific, Inc.	Accuracy: ± 0.4 ^{o}C from +5 to +40 ^{o}C, and ± 0.9^{0}C from -40^{0}C to +70^{0}C
Relative Humidity	CS215 temperature and relative humidity sensor Campbell Scientific, Inc.	Accuracy: ± 2% at 25 ^{o}C in the 10-90% range and ± 4% in the 0-100% range
Precipitation	TE25 tipping bucket rain gage with 20.32 cm orifice Texas Electronics	Accuracy: ± 1% up to 2.54 cm h^{-1}, resolution: 0.254 mm
Wind speed	R. M. Young Wind Sentry Set	Accuracy: ± 0.5 m s^{-1}
Wind direction	R. M. Young Wind Sentry Set	Accuracy: ± 0.5^{o}
Solar radiation	PYR-P Silicon Pyranometer Apogee Instruments, Inc.	Accuracy: 5% for daily total radiation
Barometric pressure	Setro 278 Barometric Pressure Sensor	Accuracy: ± 0.5 mb at +20 ^{o}C
Snow Depth	CS SR50A Sonic Ranging Sensor	± 1 cm or 0.4% of distance to target (whichever is greatest)
Soil Moisture	Stevens Hydra Probe	±0.01 WFV for most soils and ±0.03 for textured soils

Figure 2.1. Little Bear River Watershed map with water quality monitoring sites

CHAPTER 3

INCORPORATING SEDIMENT PARTICLE CHARACTERISTICS IN

TURBIDITY-BASED TSS ESTIMATION MODELS

Abstract

In the United States, suspended sediment is cited as the most common impairment to water quality. In addition to suspended sediment being a primary pollutant of concern, many other contaminants and nutrients can be associated with suspended sediment particles. Suspended sediment has significant ecological impacts, and the ability to accurately quantify suspended sediment concentrations and fluxes at appropriate temporal and spatial scales is critical in assessing whether streams are meeting their designated beneficial uses and in implementing and evaluating watershed management and mitigation plans and restoration efforts. In this study, turbidity was used as a surrogate for estimating total suspended solids (TSS) concentration at six locations in the Little Bear River Watershed, Utah, U.S.A, using linear regression techniques. At three sites, the potential effects of sediment particle characteristics on the relationships between turbidity and TSS were investigated in efforts to determine why these relationships are site specific. Findings highlight that at some monitoring locations with wide spatial distribution, the turbidity-TSS relationships are similar for more frequent (90th percentile) but lower (<45 NTU) turbidity values. Results from the particle characteristic analysis were able to inform the development of regression equations and added predictive power to the regression estimated TSS concentrations. Results showed that particle size was a

statistically significant (*p*-value<0.004) predictor of TSS and reduced overall model error from 17% to 11%.

3.1 Introduction

Monitoring programs for suspended sediment loads and concentrations are experiencing a growing need as water quality becomes an increasingly important element in the management of water resources. Although only five states in the U.S.A have established numeric water quality criteria, suspended sediment (including all solid-phase particulates) is cited as the most common impairment to water quality (U.S. EPA, 2008). Problems associated with sediment include potential impacts to: drinking water treatment processes (AWWA, 1990); fate and transport of heavy metals (Carter et al., 2006; Schoellhamer et al., 2007); recreational uses, reservoir storage and hydraulic operations (Pruitt, 2003); barge transportation and navigation lanes (U.S. Army Corps of Engineers, 2011); light suppression effects on stream bed vegetation; ecological function of aquatic habitat (Henley et al., 2000); and food webs and fish spawning beds (Rubin, 1995; Soulsby et al., 2001). Other sediment-associated impacts include the transport and fate of metals and nutrients as they are carried through river systems and are deposited and resuspended over time (Owens et al., 2001; Horowitz, 2008).

The ability to accurately quantify suspended sediment concentrations and fluxes at appropriate temporal and spatial scales is critical to assessing whether streams are meeting their designated beneficial uses and in implementing and evaluating watershed management and pollution mitigation plans. Techniques for quantifying suspended sediment concentrations have typically relied on traditional grab sampling methods,

which lack sufficient temporal frequency to capture daily, weekly and monthly fluctuations in these concentrations and fluxes (Christensen et al., 2002; Tomlinson and De Carlo, 2003; Kirchner et al., 2004; Horsburgh et al., 2010; Jones et al., 2010). More recently, methods for quantifying suspended sediment concentrations and fluxes have been developed that use high frequency, *in situ* measurements of surrogates such as turbidity to make high frequency estimates of suspended sediment concentration. Statistical regression techniques are used to relate the *in situ* turbidity measurements and much lower frequency grab samples of suspended sediment (Schoellhamer and Wright, 2003; Rasumussen et al., 2009; Jastram et al., 2010), and a resulting model is used to transform high-frequency surrogate measurements into high-frequency estimates of suspended sediment concentrations.

Turbidity has been well established as a surrogate measure of fluvial suspended sediment concentration in rivers throughout the world and is the most common surrogate used for measuring suspended sediment concentration in the United States (Gippel, 1995; Ziegler, 2002; Gray and Glysson, 2003; Pfannkuche and Schmidt, 2003; Carter et al., 2006; Stubblefield et al., 2007; Minella et al., 2008; Jastram et al., 2010; Jones et al., 2010). Although the use of surrogate methods is becoming more widespread, there are some limitations to the approach. Because turbidity is an optical measurement of the scattering of light as it passes through a given water sample, turbidity-based suspended sediment concentration estimates can be affected by particle characteristics such as size, density, organic composition, and water color that affect the measured turbidity response and its correlation with suspended sediment concentration (Gray and Gartner, 2009;

Jastram et al., 2010). Turbidity sensor output has been shown to vary by a factor of four for samples with identical suspended sediment concentrations but different particle characteristics (Gippel, 1995; Schoellhamer and Wright, 2003). It is hypothesized that this is one of the main reasons that relationships between turbidity and suspended sediment concentration are site specific. Accordingly, in this study we investigated the relationships between particle characteristics (i.e., size, concentration and surface area) and turbidity and suspended sediment concentration at three continuous monitoring sites in the Little Bear River, Utah U.S.A.

The objectives of this study were to assess the generality of using turbidity as a surrogate for suspended sediment concentration at several sites within a single watershed and to identify the physical properties of suspended sediment that contribute to the site specific nature of relationships between turbidity and suspended sediment concentration. Another objective was to investigate whether accounting for particle characteristics in the regression equations improves estimates of suspended sediment concentrations. Improved quantification of the magnitude and timing of suspended sediment loading will better inform scientific understanding of watershed response and aid in watershed management efforts such as targeting implementation of conservation and restoration efforts.

Section 3.2 of this paper provides a description of the Little Bear River watershed and water quality monitoring network. Section 3.3 describes the data collection procedures, the instrumentation for sediment particle analysis and the statistical methods used to develop surrogate relationships at six locations in the watershed. Section 3.4

describes the final surrogate equations and identifies similarities among the different sites in the watershed. Section 3.4 also demonstrates how sediment particle properties, such as size and volumetric concentration contribute to the site specificity between two sites with vast differences in spatial proximity, hydrology and water quality; and how the incorporation of these sediment properties in the regression equations improve regression estimates of suspended sediment concentration.

3.2 Study Area

The Little Bear River watershed is located in the south end of Cache Valley, Utah and encompasses roughly 740 km^2. The headwaters form in the Bear River Mountain Range, and the river drains to Cutler Reservoir on the west side of Cache Valley. Elevations within the watershed range from 2865 m to 1345 m. Land use within the upper watershed is comprised mostly of forest and range (grazing) lands while the lower portion of the watershed is made up primarily of agricultural lands and some urban development. Hydrology of the watershed is dominated by spring snowmelt, with water in the upper watershed being stored within two reservoirs, Porcupine Reservoir in the upper East Fork and Hyrum Reservoir on the mainstem, before being released and diverted into irrigation canals for agricultural use during the irrigation season.

Since 2005, research conducted by the Utah Water Research Laboratory at Utah State University has facilitated the installation of seven continuous water quality monitoring sites (Figure 3.1). Table 3.1 summarizes the locations and descriptions of the seven water quality monitoring sites located in the watershed, along with the dates from which the respective measured variables are available. Five of these sites (sites 1-5) are

located in the upper watershed, above Hyrum reservoir, where stream channels are

relatively steeper and sometimes have velocities as high as 2.0 m/s. The lower portion of

the watershed includes two monitoring sites, one below Hyrum Dam and one at the outlet

of the watershed near the confluence with Cutler Reservoir. In this lower section of the

river, channel slopes are lower and velocities are almost always <0.7 m/s. The river

exhibits a more meandering planform, and soils are predominantly fine-grained lacustrine

sediments.

3.3 Methods

3.3.1 Instrumentation and Data Acquisition

Turbidity, water temperature and gage height were measured using permanently

installed, *in situ* sensors every 30 minutes in the Little Bear River. Turbidity and water

temperature were measured using a Forest Technology Systems DTS-12 turbidity sensor.

The DTS-12 uses a laser diode with a wavelength in the near-infrared (780 nm) spectrum

and measures light scatter at 90 degrees to the incident beam (Forest Technology Systems

Ltd., 2010). This geometry results in measurement units of Nephelometric Turbidity

Units (NTU), and the probe has a range from 0-1600 NTU. Each turbidity measurement

consists of 100 instantaneous observations made over a five second time interval, after

which summary statistics are output. The median turbidity values output from the sensor

were used in this study. Temperature was measured with the DTS-12 from an internal

encapsulated thermistor and used to compensate turbidity measurements.

In situ turbidity values were paired by date and time with grab samples of TSS

and particle size distribution (PSD). Where turbidity measurements did not coincide in

time with sample collection, a turbidity estimate for the time of the TSS sample was generated by interpolating between the two turbidity measurements bracketing the sample in time. In some instances, lower turbidity values (<40 NTU) corresponded with higher TSS concentrations (>100 mg/L) or vice versa. Such anomalous values occurred in less than 5% of the total samples per site and were removed prior to developing the regression equations. Table 3.3 gives a summary of the full dataset for paired turbidity and TSS data.

Grab samples of TSS and PSD were collected at sites 2, 4 and 6 (Figure 1) because their locations represent hydrologic and water quality conditions that are of interest (i.e., they are influenced by distinct watershed characteristics described above). Generally, grab samples were collected during routine sensor cleaning and calibration visits by dipping a sample bottle or bucket approximately 30 cm below the surface of the water adjacent to the fixed turbidity sensor. Grab samples were also collected during times of elevated turbidity in order to increase the number of paired turbidity-TSS samples across a more complete turbidity range. Total suspended solids (TSS) grab samples were analyzed at the Environmental Quality Lab at the Utah Water Research Laboratory using Standard Methods 2540D, Total Suspended Solids Dried at 103-105° C (APHA, 1995). Where laboratory duplicates of TSS samples were performed, the average value of the two results was used.

Particle size and concentration were measured using a LISST-Portable (Laser In-Situ Scattering and Transmissometry) from Sequoia Scientific, which employs laser diffraction (small-angle forward laser light scattering) techniques described in AWWA

Standard No. 2560D and ISO-13320-1 (Sequoia Scientific, 2010). The particle size range of the LISST-Portable (Type C) is 1.9 – 381 μm using the randomly shaped particle inversion method (Agrawal and Pottsmith, 2000). With this model of the LISST, there are 32 size ranges logarithmically placed from 1.9 – 381 μm in which each particle is classified. The LISST-Portable outputs metrics such as volumetric particle concentration (μL/L) for each of the 32 size classes, total particle concentration (μL/L), total surface area (cm^2), mean particle diameter (μm), standard deviation and optical transmission (%).

The times at which grab samples were collected did not always correspond to elevated turbidity levels that were needed to analyze for particle characteristics with the LISST-Portable. A sufficient supply of sediment was needed for the LISST-Portable to distinguish between background noise and actual suspended sediment. Prior experiments revealed that a grab sample of 1000 mL required turbidity levels to be at least 20-30 NTU before reliable results could be obtained. For classification of particles into clay, silt and sand fractions, this research used standard particle size classifications where approximate size breaks were based on nominal diameters as follows: clay particles are <4 μm, silt 4-63 μm and sand 63-381 μm (Chapra, 1997). Additional details on sample collection and preparation and laboratory analysis can be found in Appendix A.

3.3.2 Model Development and Statistical Methods

The first procedure in model development was to determine which predictor variables would be used to model TSS concentration. Each of the potential predictor variables were plotted against the response variable (TSS) and visually examined in a scatter plot matrix to identify potential relationships, linearity and unusual or extreme

values. To accompany these plots, the Pearson's r statistic was used to report the strength

of a linear relationship between two variables. The Pearson's r is between -1 and 1 with

negative one being a perfect negative correlation and a positive one being a perfect

positive correlation.

Regression models developed in this research followed the general form:

$$y_i = \alpha_0 + \alpha_1 x_i + e_i$$ Eqn. 3.1

where y is the response variable, x_i is the predictor variable, α_o, and α_1 are regression

parameters estimated by the regression analysis, e is the error associated with each

estimate of y, and i is an index for the number of observations where $i = 1,2,…n$. The

optimum values of the parameter estimates were estimated using the ordinary least

squares (OLS) method within the R statistical computing software (http://www.r-

project.org/). All parameters used in the regression equations were tested for significance

by comparing the regression computed p-value to $\alpha=0.05$. If the p-value was less than

0.05 the parameter estimate was considered to be a significant predictor of TSS and was

used in the regression equation. The predictor variables evaluated in this study were

turbidity, turbidity squared, discharge and a categorical variable. Equation 3.2 modifies

the general form of Equation 3.1 to demonstrate the regression model used with a

categorical variable where Z is 1 when a specific condition has been met and 0 otherwise.

$$y_i = \alpha_0 + b_0 Z + \alpha_1 x_i + b_1 Z x_i + e_i$$ Eqn. 3.2

This form of the regression equation was used to simultaneously fit a model to the data in

both categories (1 and 0) and test whether their intercepts and slopes were actually

different enough to justify the addition of the categorical variable (Berthouex and Brown,

2002). The conditions for which the categorical variable was investigated at sites 1- 4 were determined using the scatter plots of turbidity and TSS which revealed where a general group of points showed a change in slope relative to the other. The categorical variable was considered in the regression analysis in order to verify that this was indeed a point of significant change in the turbidity-TSS relationship. After the general area of the slope change was identified using the scatter plot, the actual value of turbidity used as the categorical breakpoint (slope change) was found by trial and error. Again the statistical significance of the categorical parameters was evaluated using the *p*-value. Other categorical variables such as time of year and day, spring runoff versus baseflow and storm event versus no storm were previously investigated using subsets of the Little Bear River data by Jones et al., (2010), and were not found to be significant predictors for TSS.

Turbidity squared was tested in the regression equations because sometimes categorical variables can cause large, abrupt shifts in estimated TSS concentration for small changes in turbidity. These dramatic shifts in estimated TSS are not likely to occur for such minute changes in turbidity and the effect of replacing the categorical variable with the turbidity squared term brought a smoothing, curvilinear fit to the data. Although in general, models were selected based on the lowest error (Eqn. 3.3), the smoothing effects of the turbidity squared term when compared to the categorical variable were also taken into consideration if the respective parameter estimate was below the significance threshold of $\alpha=0.05$.

In linear regression, model fit is generally contingent upon the following four assumptions about the response variable y_i: errors are independent, normally distributed,

have constant variance, and that the true relationship between the response and predictor variables is linear. These assumptions were validated by inspecting graphical plots of the residual (observed – predicted TSS) values. These plots demonstrated that because the residuals were normally distributed, independent, have constant variance and a mean of zero, the assumptions of linear regression were met and the corresponding model fit is a valid representation of the data. Although the results of the residual analysis are not formally discussed here, Appendix B provides the residual analysis for each site. Appendix B also demonstrates the results of using variety of transformations and weighted least squares (site 4). Both of these techniques, however, were not successful at improving the characteristics of normality, independence, linearity and homoscedasticity (constant variance).

Alternate models were evaluated based on the root mean square error (RMSE), (Equation 3.3), which is a measure of the overall error associated with a given model.

$$RMSE = \sqrt{\frac{\sum e_i}{v}} \qquad \qquad \text{Eqn. 3.3}$$

The error term e_i is the difference between the observed TSS and the model predicted TSS for each observation (i), and v is the degrees of freedom from the regression model. The lower the RMSE, the closer the predicted response will be (on average) to the observed TSS value. The RMSE can also be expressed as a percentage interval using Equation 3.4:

$$SPE = \pm\frac{RMSE}{\bar{y}} * 100 \qquad \qquad \text{Eqn. 3.4}$$

where $\bar{y}$ is the mean observed TSS concentration. This standardized percentage error (SPE) was also used because it expresses model uncertainty in terms of an interval and a

percent that is comparable across models and monitoring sites. In general, the model

with the lowest RMSE was selected as the final model. F-tests were conducted to

compare the variance in regression-computed TSS concentrations between selected sites.

The test statistic is a ratio of the variance of the estimated TSS concentrations from one

site to a second as shown in the following equation:

$$F = \frac{S_a^2}{S_b^2}$$
<div align="right">Eqn. 3.5</div>

where S_a is the variance of site a and S_b is the variance of site b. Equation 3.6 shows the

calculation used for variance

$$S^2 = \frac{1}{n-1}\sum_{i=1}^{n}(X_i - \bar{X})^2$$
<div align="right">Eqn. 3.6</div>

where X is estimated TSS concentration, $\bar{\mathrm{x}}$ is the average observed TSS concentration

and n is the number of observations with n-1 degrees of freedom. The null hypothesis is

that the variances are equal. The null hypothesis is rejected when the F-ratio is greater

than the critical value in the F-table, for the given degrees of freedom and a significance

level of $\alpha = 0.05$.

The second procedure in model development (for sites 2, 4 and 6) was to use a

subset of the full turbidity and TSS data (Table 3.3) to investigate which sediment

particle characteristics might be significant predictors of TSS. For this analysis, the

subset of data included only paired TSS, turbidity and particle characteristics samples

collected from January to May 2012. The subset of data contained 11-12 data values,

while the full set of data, which was specific to each site, contained the count values

found in Table 3.3. Particle characteristics investigated included total sample volumetric

particle concentration (μL/L), particle surface area (cm^2) and volumetric particle

concentration (as μL/L and as a percent of total concentration) in each of the 32 size

classes. Additionally, specific groups of particle size bins were determined and analyzed

as individual predictors. These groups included size ranges for clay, silt and sand as well

as a group for the portion <63 μm and a group for the portion >63 μm. The groups above

and below 63 μm were investigated because others have noted that simple linear

regression of turbidity and TSS is most suited for situations where the sand-size (> 63

μm) portion of the samples is less than 20% (Rasmussen et al., 2009).

Once a particular particle characteristic was found to be a statistically significant

(p-value <0.05) predictor of TSS for the subset of data, additional values of those particle

characteristics were then modeled using the regression techniques described with stream

discharge as the predictor so that values of the particle characteristic could be computed

and augmented to the full set of data. This made it possible to test the significance of

particle characteristics on the same scale as other continuously monitored variables such

as turbidity.

3.4 Results and Discussion

3.4.1 Model Development

A summary of the TSS, turbidity and discharge data used to develop the

correlation matrices and surrogate relationships is shown in Table 3.3. Figures C3.1

through C3.6 in Appendix C show correlation matrix plots for each site that had four or

more of the following variables: TSS, turbidity, discharge, turbidity squared, categorical

variable (Z) and particle size. The correlation matrices show a high correlation between

TSS and turbidity where the Pearson's r correlation coefficient is greater than or equal to 0.86 for each of the six sites. The correlation coefficients between TSS and discharge are lower ranging from 0.29-0.66. For TSS and turbidity squared, the correlation coefficient is 0.94 and greater for sites 1, 2, 3, and 4. The correlation coefficients for turbidity squared term at sites 5 and 6 were 0.85 and 0.61, respectively.

The categorical variable was found to be a significant predictor of TSS concentration at sites 1-4 with all p-values <0.0006 as shown in Table 3.4. The categorical breakpoint for turbidity at sites 1, 2, and 3 was 40 NTU. For site 4, the categorical breakpoint for turbidity was 600 NTU. For sites 1-4, the categorical variable introduced an artificial shift in the estimated TSS concentration for small changes in turbidity. When turbidity increases by 1 NTU at the categorical breakpoints, estimated TSS concentrations shift by -3.4, +1.3, -12.0, and -279 mg/L for sites 1-4, respectively.

The turbidity squared term—along with turbidity—was a significant (all p-values <0.001) predictor of TSS for sites 1-4, but the model containing the categorical variable for these sites gave a better fit with a greater reduction in overall model error. For site 4, the reduction in overall error from the model with turbidity only to the model with turbidity and the categorical variable was from 63 mg/L to 48 mg/L, with the standardized percentage error being reduced from ±34% to ±26%. As previously stated, a change of 1 NTU at the categorical breakpoint of 600 NTU at site 4 causes an artificial drop in estimated TSS concentration of 279 mg/L (1015mg/L to 736mg/L) or 27%, but because the model with the categorical variable causes a significant reduction in model error, it was selected as the final model. For site 5, turbidity was the only significant

predictor in the regression equation (Table 3.4) for which data included TSS values up to 27 mg/L. Sediment sample data for this site was limited because events that cause higher suspended sediment concentrations are controlled by dam releases from Hyrum reservoir. These events are difficult to anticipate because they do not always correspond with conditions at other sites in the watershed. For site 6, TSS concentration was modeled best using turbidity and turbidity squared (p-value $\ll 0.001$) where the RMSE of the turbidity only model was reduced from 10.04 to 8.40 mg/L (SPE reduction of 5%). For each site, the respective parameter estimates, p-values, RMSE and SPE for final models are shown in Table 3.4.

Additionally, discharge was tested for significance as a predictor in each of the models. For sites 1-4, discharge—along with turbidity—was a significant (p-value <0.05) predictor of TSS, but the categorical variable proved to be a better (lower p-value and RMSE) predictor of TSS than discharge when discharge and the categorical variable were tested separately. When tested together discharge provided no significant reduction in error (RSME) and the simpler model was preferred. For sites 5 and 6, discharge was not a significant predictor with turbidity (or turbidity and turbidity squared for site 6) in estimating TSS concentration.

3.4.2 Comparison of Regression Equations

Figure 3.2 shows individual scatter plots of TSS and turbidity for each site with their respective modeled regression lines. The four upper watershed sites (panels A-D) exhibit regressions that are linear and steep, with slopes greater than one (Table 3.4). Sites 1 and 2 display the regression models for turbidities up to ~560 NTU and sites 3

and 4 up to 92 NTU and 1407 NTU, respectively. Slopes greater than one suggest higher

concentrations of TSS per unit of turbidity resulting from larger and/or denser particles.

Site 5 in the lower watershed had a slope slightly greater than 1 that follows closely the

1:1 relationship shown in Figure 3.2 (panel E) for values of turbidity up to 27 NTU.

Panel F for site 6 shows a curvilinear fit for the data where TSS and turbidity increase

almost equally until ~100 NTU where the slope decreases significantly with TSS as

turbidity continues to increase reaching 166 NTU. The curvilinear fit appears to be

greatly influenced by the two high (139 and 166 NTU) turbidity values that seem

anomalous when compared to the grouping of the remaining 239 paired turbidity-TSS

values. These two turbidity values are indeed rare, as less than 0.1% of all turbidity

values measured at this site are greater than 100 NTU. The curvilinear fit however, is

still suggested by the data (and by particle characteristics yet to be discussed) because the

turbidity squared term is statistically significant (p-value $<<0.001$) and the overall error is

reduced compared to the turbidity only model even when those two points are removed

from the regression analysis. The turbidity squared however is limited as turbidity values

greater than 149 NTU would cause the TSS estimate to decrease. Although this model is

only valid for turbidities equal or less than 149 NTU, 149 NTU is the 99.95th percentile

of more than 114,000 observed turbidity values.

While it is generally accepted that turbidity-TSS relationships must be developed

uniquely for each particular location (Riley, 1998; Gray et al., 2003; Schoellhamer and

Wright, 2003; Rasmussen et al., 2009), this research shows that several turbidity-TSS

relationships in the Little Bear River watershed are quite similar (e.g., for a turbidity of

30 NTU their equations give very similar TSS concentrations) in their ability to estimate

TSS concentration for the more frequent but lower turbidity values. Figure 3.3 shows

each of the regression lines at the six sites for turbidity values 0-200 NTU. Similarity

among the surrogate relationships at sites 2-6 is observed as each of these regression lines

lay close together above the 1:1 line until ~40 NTU, the categorical breakpoint for sites

1-3. Sites 2 and 3 are very similar in this range with maximum differences in estimated

TSS being ~5 mg/L. Sites 2 and 6 are also very much the same in this range with

maximum differences in estimated TSS being <5 mg/L before the regression line of site 6

dramatically decreases in slope, diverging from sites 2, 3, and 4 starting at ~60 NTU.

Sites 2 and 4 are also very similar in estimated TSS where the difference between the two

is a maximum of 22 mg/L when turbidity reaches 200 NTU. The turbidity-TSS model

for site 5 was developed for turbidity values up to 27 NTU. In this range, site 5 is similar

to sites 2, 3, and 6 and nearly the same as site 4 with a max difference of <13mg/L. As

shown in Figure 3.3, the regression line for site 1 is the most withdrawn from the others

with a slope distinctly less than 1 until it reaches its categorical breakpoint of 40 NTU,

where it increases and begins a steep climb crossing the 1:1, but maintaining a clear

separation (but similar slope) of >30mg/L from all other sites up to 200 NTU.

The similarity in TSS concentrations (for turbidities <40 NTU) at sites 2, 4, and 6

are further demonstrated in Table 3.7. The F-ratios are 1.8 for Site 4 / Site 2, 1.2 for Site

6 / Site 2, and 1.5 for Site 4 / Site 6 and all are less than the critical value, which for all

cases is 1.9. Because the ratios of the variances are less than the critical F values, there is

no evidence to reject the null hypothesis in any of the cases tested.

The ranges of turbidity for which the sites have been compared is important to highlight because the 1-40 NTU range comprises 90% of the >77,000 observed turbidity values for which sites 2-6 are similar; the 1-200 NTU range covers ~99% of the ~84,500 and ~95% of the >118,000 observed turbidity values for sites 2 and 4, respectively. These results suggest that for the majority of turbidity measurements made throughout the year at sites 2-6, a single, general TSS regression model could be used to estimate TSS concentrations and fluxes.

Similarities among these regression equations are marked by their ability to predict TSS for a given range of turbidity. Differences in predicted TSS concentrations between the aforementioned sites were small enough that they were within the expected measurement error. This is based on the level of precision reported with the laboratory results of TSS on duplicate samples at the Utah Water Research Laboratory and the level of precision reported by Standard Methods 2540 D, total suspended solids determination (APHA, 1995). Laboratory results reported standard deviations of 2-14 mg/L for a TSS range of 10-90 mg/L and standard deviations of 25-35 mg/L for TSS values in the range of 120-720 mg/L. Standard Methods 2540 D gives a standard deviation of 5.2 mg/L of TSS for a known level of 15 mg/L and a standard deviation of 24 mg/L of TSS for a known level of 242 mg/L. The levels of precision reported by both sources agree with one another for their respective TSS ranges.

3.4.3 Contribution of Sediment Particle Characteristics

As shown in figure 3.3, the regression computed TSS concentrations for sites 2, 4 and 6 would be similar up to the 40-60 NTU range, where site 6 diverges from sites 2 and

4. The reason for the departure of site 6 is found within an analysis of the particle size distributions and particle concentrations conducted at these sites.   As shown in Figure 3.3, the regression line for site 6 lays very close to the regression lines for sites 2-4 until about ~60 NTU where the slope begins to decrease causing turbidity to rise disproportionately more than TSS, resulting in a wide separation from the regression lines of sites 2 and 4. Figure 3.4 demonstrates how particle size distributions at site 6 take on a distinct change at the higher turbidity levels. In Figure 3.4 the particle size distributions for site 6 are displayed with percent total volumetric concentration on the y-axis and a log scale of particle size (1-350 μm) on the x-axis. The solid black and grey lines represent the particle distributions for turbidity values of 139 and 166 NTU and the dashed lines represent turbidity values of 38 and 41 NTU. The solid dark lines mark a very distinct and different pattern from the size distributions of the lower turbidities, demonstrating that not only were the particles smaller (on average), but there were greater numbers of particles overall (increased particle concentrations), particularly in the smaller (1-20 μm) sized bins. This shift toward the smaller particles for higher turbidities demonstrated in Figure 3.4 is believed to be the cause of the decreasing slope of the curvilinear regression equation because the smaller particles at this site would have less mass while still increasing turbidity. It is true that even if each particle has less mass, if there are more particles the overall mass could still be higher. But because the particle mass at site 6 does not increase with turbidity as the other sites do, the curvilinear regression equation is suggested.

This logic is reasonable for several reasons. First, the location of site 6 is below Hyrum Reservoir, which prevents larger sediment particles in the upper watershed from reaching the lower watershed site. Sediment sources contributing to site 6 are limited to the surrounding land, which is made up of finer silts and clays. Second, lower channel gradients cause water velocities to be low enough that even when discharge is at its height, stream velocities are not high enough to carry larger, denser particles in the water column. These factors may explain the decrease in slope for the curvilinear regression equation that causes the turbidity-TSS relationship at site 6 to deviate from the upper watershed sites.

The particle size distributions for sites 2, 4, and 6 also provide insight into why the regression estimated TSS concentrations are similar for the lower turbidity values. Figure 3.5 demonstrates two particle size distributions for each of the three sites where turbidities are between 35 and 49 NTU. The quantity of particles for a given distribution is tabulated in Table 3.5 (as particle concentration) and as a table within the plot. At these low turbidity values, their respective particle size distributions are very similar. The distribution for site 6 varies slightly from the other distributions as it has more particles in the 10-20 μm range and fewer particles in the 50-300 μm range when compared to site 2 and 4. Although particle quantities vary among sites, the shape of the particle size distributions follow the same general pattern. Because particle concentration varies greatly among these distributions for the same turbidity values, it is believed that particle size has the greater impact on turbidity, causing the similarities between these sites.

In contrast with site 6, where particle size, quantity, and density influenced the turbidity-TSS relationship at higher turbidities, the particle size distributions at site 2 for higher turbidities suggested particle concentration influences the turbidity-TSS relationship more. Figure 3.6 shows four particle size distributions for site 2 at low (24-46 NTU) and high (82 and 143 NTU) turbidities. The particle size distributions have the same patterns for both low and high turbidity values. This distribution is in stark contrast to site 6 where particle size was more variable as turbidity increases, especially in the smaller (1-10 μm) particle size range. The more consistent distributions at site 2 suggest that the turbidity-TSS relationship is influenced more by particle concentration than size. This is shown by the higher turbidity levels also having significantly higher particle concentrations (88-90%) and particle mass (44-92%). Looking back at site 6, the particle concentrations also increased by a similar (56-76%) amount when turbidities changed from low to high. This suggests that when both sites increase to similar levels of turbidity and particle concentration (still considering their respective particle size distributions), the major factors affecting the surrogate relationships are particle concentrations (quantity) at site 2 and particle size at site 6. For reasons given earlier of why particle densities at site 6 would remain low at higher turbidities, the opposite is true at site 2 for larger particles during higher turbidity levels. At site 2, channel gradients have steeper slopes resulting in higher water velocities that enable the stream to suspend larger, and potentially heavier particles in the water column.

Finally, an attempt was made to correlate sediment particle characteristics and turbidity with TSS. For sites 2, 4, and 6 a subset of 12 paired turbidity and TSS values

was created and matched with percent of total (volumetric) concentration (PTC) for each of the 32 size bins, overall sample (volumetric) concentration and particle surface area. For sites 4 and 6, several of the smaller particle size bins of PTC—along with turbidity—were statistically significant (p-value <0.05) predictors of TSS. For site 2, only the smallest PTC size bin of 1.9-2.25 μm showed a significant correlation to TSS concentration. This size bin represented, on average (across samples), approximately 1.1% of the overall sample volumetric concentration of particles and was a statistically significant (p-value <0.01 level) predictor with turbidity in estimating TSS concentration for the subset of data at site 2.

At sites 4 and 6, discharge was a statistically significant (p-value<0.05) predictor of some of the smaller PTC size bins, but once the resulting PTC values—calculated from the discharge-PTC regression—for these size bins were augmented to the full dataset, their respective PTC size bins were not statistically significant predictors with turbidity to model TSS concentration. At site 2, discharge was a statistically significant (p-value <0.01) predictor for the 1.9-2.25 μm PTC size bin. Using the full data set at site 2, turbidity and the PTC of the 1.9-2.25 μm size bin were found to be statistically significant (p-value <<0.001) predictors of TSS concentration.

The discharge-PTC regression equation used to calculate the PTC values at site 2 for the 1.9-2.25 μm size bin is shown in Table 3.6. Also shown in Table 3.6 are the summary statistics for the model that used turbidity and the PTC size bin of 1.9-2.25 μm to estimate TSS concentration as well as the turbidity only model. When compared to the turbidity only model, the RMSE was reduced from 14.6 to 13.5 mg/L with overall

error being reduced by 1.7% as shown in Table 3.6. Although the PTC size bin of 1.9-
2.25 μm was less significant when compared to the categorical variable of the model
shown in Table 3.4, the significance of this particle characteristic as a predictor in the
regression analysis and the reduction in error demonstrate that accounting for particle
characteristics in the regression equations has the potential to improve estimates of TSS
concentration.

The results indicate that sediment particle characteristics can provide two benefits
to water quality monitoring programs that use turbidity as a surrogate for suspended
sediment concentrations. First, sediment particle characteristics can inform the
development of regression equations and are helpful in explaining why some sites have
regression models that are significantly different than others. Specifically, this
understanding informs the development of the surrogate relationship at site 6. For
example, one could build the regression by omitting the two highest turbidity values
(calling them anomalous) and complete the regression with a simple straight line through
the data points. This, however, would assume an constant rate of change for TSS and
might severely overestimate TSS concentrations at this location when turbidities
exceed100 NTU. However, the particle characteristic data suggest that TSS
concentrations do not increase in a 1:1 fashion. Second, the sediment particle
characteristic data are also able to improve the surrogate relationships at some sites by
adding predictive power to the estimates of TSS concentration. This was demonstrated at
site 2 where the percent of particles within the smallest particle size range was a

significant predictor with turbidity in estimating TSS concentration and reducing overall model error.

3.5 Conclusions

Regression models to predict TSS concentrations were developed at six locations in the Little Bear River watershed. Turbidity and a categorical variable indicating whether turbidity was above or below 40 NTU were found to be significant predictors for TSS concentration at the four locations (sites 1-4) in the upper watershed. Of the two sites (5 and 6) in the lower watershed, turbidity and turbidity squared were significant predictors of TSS concentration at site 6, but at site 5 only turbidity was significant. The results show that for the overall turbidity range at each site, the regression equations are site specific. The results, however, also show that for sites 2 and 4-6 the regression equations are very similar for the lower (<40 NTU) turbidity range. At these lower levels of turbidity the differences between estimated TSS concentrations among the four sites is negligible.

Particle size analysis demonstrated that turbidity values less than 40 NTU resulted in similar particle size distributions among sites, even though particle concentrations were not similar. Particle size analysis also showed that as turbidities surpassed ~40-60 NTU, particle size distributions where distinctly different among the sites. These results suggest that the dominate particle property influencing the turbidity-TSS relationship changes among sites.

The physical attributes (steeper channel gradients and higher stream velocities at site 2; low-slope channel gradients and lower stream velocities at site 6, aide in this

evaluation), and related particle characteristics demonstrated in these results, also suggested a connection to the regression model developed at site 6 where the turbidity squared term was found to be a significant predictor with turbidity of TSS concentration. These results suggest that sediment particle analysis can provide valuable information in the development of the site specific regression equations.

Results from the particle characteristic analysis show that the quantity of particles in the smallest size bin (1.9-2.25 μm) can be incorporated into regression models to reduce the RMSE. The results show that because fewer, smaller particles pass through site 2, percent concentration (PTC) values for the 1.9-2.25 μm size bin were significant with turbidity in predicting TSS concentration on the full data set.

This research has important application to sediment monitoring programs in rivers and streams where sediment related impairments occur. Surrogate relationships can be enhanced with the use of particle characteristics data to improve estimates of TSS concentrations, which can then be used to assess exceedance of water quality criteria and help better understand the magnitude and timing of sediment loads. This information can help in determining whether rivers and stream are meeting their beneficial use and in targeting the implementation of conservation and restoration efforts.

References

Agrawal, Y. C., and Pottsmith, H. C. (2000). "Instruments for particle size and settling velocity observations in sediment transport." *Marine Geology*, 168(1-4), 89-114.

American Public Health Association (APHA). (1995). "Standard methods for the examination of water and wastewater." American Public Health Association, American Water Works Association, and Water Pollution Control Federation, Washington, DC.

American Water Works Association (AWWA). (1990). *Water quality and treatment: A handbook of community water supplies*. McGraw-Hill, New York.

Anderson, C. W. (2005). "Turbidity 6.7." U.S. Geological Survey Techniques of Water-Resources Investigations, book 9, chap. A6., sec. 6.7. <http://pubs.water.usgs.gov/twri9A6/> (1 October 2011).

Berthouex, P. M., and Brown, L. C. (2002). *Statistics for environmental engineers. Lewis Publishers*, New York, 489.

Carter, J., Walling, D. E., Owens, P. N., and Leeks, G. J. L. (2006). "Spatial and temporal variability in the concentration and speciation of metals in suspended sediment transported by the River Aire, Yorkshire, UK." *Hydrological Processes*, 20(14), 3007-3027.

Chapra, S. C. (1997). *Surface water-quality modeling*. McGraw-Hill, New York.

Christensen, V., Rasmussen, P., and Ziegler, A. (2002). "Real-time water quality monitoring and regression analysis to estimate nutrient and bacteria concentrations in Kansas streams." *Water Science and Technology*, <http://ks.water.usgs.gov/pubs/reports/vgc.0610.html> (July 15, 2011).

Forest Technology Systems, Ltd. (2010). DTS-12 SDI-12 Turbidity Sensor Operating Manual. Forest Technology Systems, Victoria, BC.

Gippel, C. J. (1995). "Potential of turbidity monitoring for measuring the transport of suspended solids in streams." *Hydrological Processes*, 9(1), 83-97.

Gray, J. R., and Glysson, G. D. (2003). "Federal Interagency Workshop on Turbidity and other Sediment Surrogates." Proceedings of the Federal Interagency Workshop on Turbidity and Other Sediment Surrogates, U.S Geological Survey, Reno, NV, April 30-May 2, 2002.

Gray, J. R., Patiño, E., Rasmussen, P. P., Larsen, M. C., Melis, T. S., Topping, D. J., and Alamo, C. (2003). "Evaluation of sediment surrogate technologies for computation of suspended-sediment transport." Proceedings of the 1st International Yellow River Forum on River Basin Management, 21-24.

Gray, J. R., and Gartner, J. W. (2009). "Technological advances in suspended-sediment surrogate monitoring." *Water Resources Research*, (November 2008).

Henley, W. F., Patterson, M. A., Neves, R. J., and Lemly, A. D. (2000). "Effects of sedimentation and turbidity on lotic food webs: A concise review for natural resource managers." *Reviews in Fisheries Science*, (August 2011), 37-41.

Horowitz, A. J. (2008). "Determining annual suspended sediment and sediment-associated trace element and nutrient fluxes." *The Science of the Total Environment*, Elsevier B.V., 400(1-3), 315-43.

Horsburgh, J. S., Spackman Jones, A., Stevens, D. K., Tarboton, D. G., and Mesner, N. O. (2010). "A sensor network for high frequency estimation of water quality constituent fluxes using surrogates." *Environmental Modeling & Software*, Elsevier Ltd, 25(9), 1031-1044.

Jastram, J. D., Zipper, C. E., Zelazny, L. W., and Hyer, K. E. (2010). "Increasing precision of turbidity-based suspended sediment concentration and load estimates." *Journal of Environmental Quality*, 1306.

Jones, A. S., Stevens, D. K., Horsburgh, J. S., and Mesner, N. O. (2010). "Surrogate measures for providing high frequency estimates of total suspended solids and total phosphorus concentrations." *Journal of the American Water Resources Association*, 47(2), 239-253.

Kirchner, J. W., Feng, X., Neal, C., and Robson, A. J. (2004). "The fine structure of water-quality dynamics: the (high-frequency) wave of the future." *Hydrological Processes*, 18(7), 1353-1359.

Minella, J. P. G., Merten, G. H., Reichert, M., and Clarke, R. T. (2008). "Estimating suspended sediment concentrations from turbidity measurements and the calibration problem." *Hydrological Processes*, 1830(August 2007), 1819-1830.

Owens, P. N., Walling, D. E., Carton, J., Meharg, A. A., Wright, J., and Leeks, G. J. L. (2001). "Downstream changes in the transport and storage of sediment-associated contaminants (P, Cr and PCBs) in agricultural and industrialized drainage basins." *The Science of the Total Environment*, 266(1-3), 177-86.

Pfannkuche, J., and Schmidt, A. (2003). "Determination of suspended particulate matter concentration from turbidity measurements: Particle size effects and calibration procedures." *Hydrological Processes*, 17(10), 1951-1963.

Pruitt, B. A. (2003). "Uses of turbidity by states and tribes." Proceedings of the Federal Interagency Workshop on Turbidity and Other Sediment Surrogates, U.S Geological Survey, Reno, NV, April 30-May 2, 2002.

Rasmussen, P. P., Gray, J. R., Glysson, G. D., and Ziegler, A. C. (2009). "Guidelines and procedures for computing time-series suspended-sediment concentrations and loads from in-stream turbidity-sensor and streamflow data." U.S. Geological Survey Techniques and Methods book 3, chap. C4, 53 p.

Riley, S. J. (1998). "The sediment concentration–turbidity relation: Its value in monitoring at Ranger Uranium Mine, Northern Territory, Australia." *Catena*, 32(1), 1-14.

Rubin, J. F. (1995). "Estimating the success of natural spawning of salmonids in streams." *Journal of Fish Biology*, Wiley Online Library, 46(4), 603-622.

Schoellhamer, D. H., Mumley, T. E., and Leatherbarrow, J. E. (2007). "Suspended sediment and sediment-associated contaminants in San Francisco Bay." *Environmental Research*, 105(1), 119-31.

Schoellhamer, D.H., and Wright, S. A. (2003). "Continuous measurement of suspended-sediment discharge in rivers by use of optical backscatterance sensors." *IAHS Publication*, (June 2002), 28-36.

Sequoia Scientific Inc. (2010). *LISST-Portable (Type C) operating manual*. Bellevue, WA.

Soulsby, C., Youngson, A. F., Moir, H. J., and Malcolm, I. A. (2001). "Fine sediment influence on salmonid spawning habitat in a lowland agricultural stream: A preliminary assessment." *The Science of the Total Environment*, 265(1-3), 295-307.

Stubblefield, A. P., Reuter, J. E., Dahlgren, R. A., and Goldman, C. R. (2007). "Use of turbidometry to characterize suspended sediment and phosphorus fluxes in the Lake Tahoe basin, California, USA." *Hydrological Processes*, Wiley Online Library, 21(3), 281-291.

Tomlinson, M. S., and De Carlo, E. H. (2003). "The need for high resolution time series data to characterize Hawaiian streams." *Journal of the American Water Resources Association*, 39(1), 113-123.

U.S. Army Corps of Engineers. (2011). "Maintenance dredging and the environment." <http://www.nab.usace.army.mil/whatwedo/dredging.htm> (Aug. 1, 2011).

U.S. Environmental Protection Agency (U.S. EPA). (2008). "National summary of impaired waters and TMDL information." <http://iaspub.epa.gov/waters10/attains_nation_cy.control?p_report_type=T#causes_303d> (Nov. 1, 2011).

Ziegler, A. C. (2002). "Issues related to use of turbidity measurements as a surrogate for suspended sediment." Proceedings of the Federal Interagency Workshop on Turbidity and Other Sediment Surrogates, U.S Geological Survey, Reno, NV, April 30-May 2, 2002.

Table 3.1. Summary of water quality monitoring sites in the LBR watershed

Site	Site Name	Date Range	Latitude	Longitude	Site Description
1	Upper South Fork	Oct. 2007-Present	41.4954	-111.818	Located in the upper portion of the watershed where land use is primarily forest and grazing
2	Lower South Fork	July 2007-Present	41.5065	-111.8151	Located below the confluence of the Upper South Fork and a major tributary, Davenport Creek
3	Avon/Confluence	Nov. 2007-Present	41.5361	-111.8305	Located below the confluence of the East Fork and Lower South Fork sites
4	Paradise	June 2005-Present	41.5756	-111.8552	Located a few miles upstream of Hyrum Reservoir
5	Wellsville	Nov. 2007-Present	41.6435	-111.9176	Located a few miles below Hyrum Reservoir, flows are controlled by dam releases and summer flows are dominated by groundwater flow
6	Mendon	August 2005-Present	41.7185	-111.9464	Located near the boundary of the watershed at Cutler Reservoir. Water quality is affected by agriculture return flows and the Wellsville WWTP

Table 3.2. Summary of water quality sensor specifications

Variable	Sensor	Specifications
Turbidity	DTS-12 turbidity sensor, Forest Technology Systems, Inc.	Accuracy: ± 2% 0-500 NTU and ± 4% 501-1600 NTU
Temperature	Encapsulated Thermistor sensor, Forest Technology Systems, Inc.	Accuracy: ±0.2^{o}C
Stage	SPXD-600 Pressure Transducer, KWK Technologies, Inc.	Accuracy: ± 1% of the full measurement span

Table 3.3. Summary statistics for field variables used to develop surrogate relationships for turbidity and total suspended solids (TSS) concentration at each of the six sites

Site	Variable	Median	Mean	Standard Deviation	Range	Count
1	Turbidity (NTU)	9.78	55.0	108	1 - 560	91
	TSS (mg/L)	10.0	79.0	181	1 - 960	91
	Discharge (m^3/s)	0.66	0.82	0.6	0.2-2.3	91
2	Turbidity (NTU)	9.13	39	88	1 - 556	138
	TSS (mg/L)	14.2	67	162	2 - 1110	138
	Discharge (m^3/s)	1.56	3.36	3.49	0.3-16.3	138
3	Turbidity (NTU)	4.98	12	17	1 - 92	68
	TSS (mg/L)	8.7	17	24	2 - 146	68
	Discharge (m^3/s)	0.93	2.79	3.27	0.1-12.2	68
4	Turbidity (NTU)	8.4	115	266	1 - 1407	136
	TSS (mg/L)	14.75	186	435	1.2 - 2450	136
	Discharge (m^3/s)	1.76	3.82	3.44	0.4-12.5	136
5	Turbidity (NTU)	2.98	5.0	5.0	0.3 - 27	80
	TSS (mg/L)	6.86	8.0	6.9	0.2 - 35	80
	Discharge (m^3/s)	1.44	2.8	3.3	0.01-13	80
6	Turbidity (NTU)	22.45	23.29	16.7	3.7 - 166	241
	TSS (mg/L)	30.0	31.2	19.6	3.3 - 92	241
	Discharge (m^3/s)	2.26	3.83	2.9	1.0-11	241

Table 3.4. Little Bear River watershed regression equations with summary statistics. The regression equations were developed using empirical field data from a continuously monitored sensor network. A summary of variables measured is tabulated in Table 3.3. Regression equations are plotted in Figure 3.3

Site	Equation	p-value	Standard Error	RMSE (mg/L)	% Error (±)
1	TSS= 0.57*Turb-52.74* Z + 1.22*Z*Turb	< 0.0004	Turb: 0.15 Z: 3.7 Z*Turb: 0.15	10.3	13.0
2	TSS= 1.28*Turb-25.31*Z +0.63 *Z*Turb	< 0.0006	Turb: 0.18 Z: 3.9 Z*Turb: 0.18	12.9	19.2
3	TSS= 3.24+1.073*Turb-55.88 *Z+1.070*Z* Turb	< 2e-07	Turb: 0.09 Z: 9.5 Z*Turb: 0.17	6.1	35.5
4	TSS= 1.70*Turb-516.12*Z+ 0.39*Z*Turb	< 2e-08	Turb: 0.04 Z: 53.8 Z*Turb:0.06	48.2	25.9
5	TSS= 2.69 + 1.12* Turb	< 8e-07	Turb: 0.067	3.2	40.6
6	TSS=1.60*Turb - 0.005* $Turb^2$	< 2e-16	Turb: 0.07 $Turb^2$: 5.1e-04	8.4	26.7

Variable Description

TSS	Total Suspended Solids (mg/L)
Turb	Turbidity (NTU)
Z	Sites 1, 2 and 3 Z=1 when Turb $\geq$ 40 NTU; Site 4 Z=1 when Turb $\geq$ 600 NTU; otherwise Z=0
$Turb^2$	Curvilinear term for turbidity (NTU)

Parameter	**Description**
p-value	A probability; represents all variables used in the equation
RMSE	Root mean square error (mg/L)
%Error	Standardized percent error (SPE) (±) for the given model

Table 3.5. Summary of turbidity, total suspended solids (TSS) concentration and volumetric particle concentration at three sites on the Little Bear River. TSS and particle concentrations are shown for low (35-46 NTU) and high (143-702 NTU) turbidity values

Site	Date	Turbidity (NTU)	TSS (mg/L)	Particle Concentration (uL/L)
2	20-Jan-12	82	126	555
2	12-Mar-12	46	70	54
2	23-Mar-12	143	246	605
2	23-Apr-12	24	19	66
4	20-Jan-12	702	720	683
4	12-Mar-12	190	657	249
4	13-Apr-12	37	83	162
4	26-Apr-12	35	49	398
6	19-Jan-12	166	137	520
6	20-Jan-12	139	99	551
6	23-Mar-12	41	66	130
6	25-Mar-12	38	54	220

Table 3.6. Comparison of regression models that incorporate sediment particle characteristics for a subset of data at site 2 in the Little Bear River Watershed with a summary of the regression statistics. The first model estimates the PTC of 1.9-2.25 μm size bin using discharge (m^3/s). In the second model PTC is used as a predictor with turbidity to predict TSS concentration. The third model uses turbidity only as a predictor

Site	Equation	p-value	Standard Error	RMSE	Standardized % Error ($\pm$)
2	$PTC_1 = 0.148 * Q$	<0.01	Q: 0.045	0.51	0.74
	TSS = 1.9 * Turb - 13.02 * PTC_1	<<0.001	Turb: 0.015 PTC: 2.6	13.5	20.0
	TSS = -4.7 + 1.8 * Turb	<<0.001	Turb: 0.014	14.6	21.7

Variable	Description
TSS	Total Suspended Solids (mg/L)
Turb	Turbidity (NTU)
PTC_1	Percent Total Concentration of the particles in the 1.9-2.25μm particle size bin
Q	Discharge (m^3/s)

Parameter	Description
p-value	Represents all variables used in the equation
RMSE	Root mean square error in units of the dependent variable
Std. % Error	Percent error ($\pm$) for the given model

Table 3.7. F-ratios for comparison of model variance

Sites Compared	F-Ratio	F-Critical
Site 4 /Site 2	1.8	1.9
Site 6/ Site 2	1.2	1.9
Site 4/ Site 6	1.5	1.9

Figure 3.1. Little Bear River watershed water quality monitoring sites

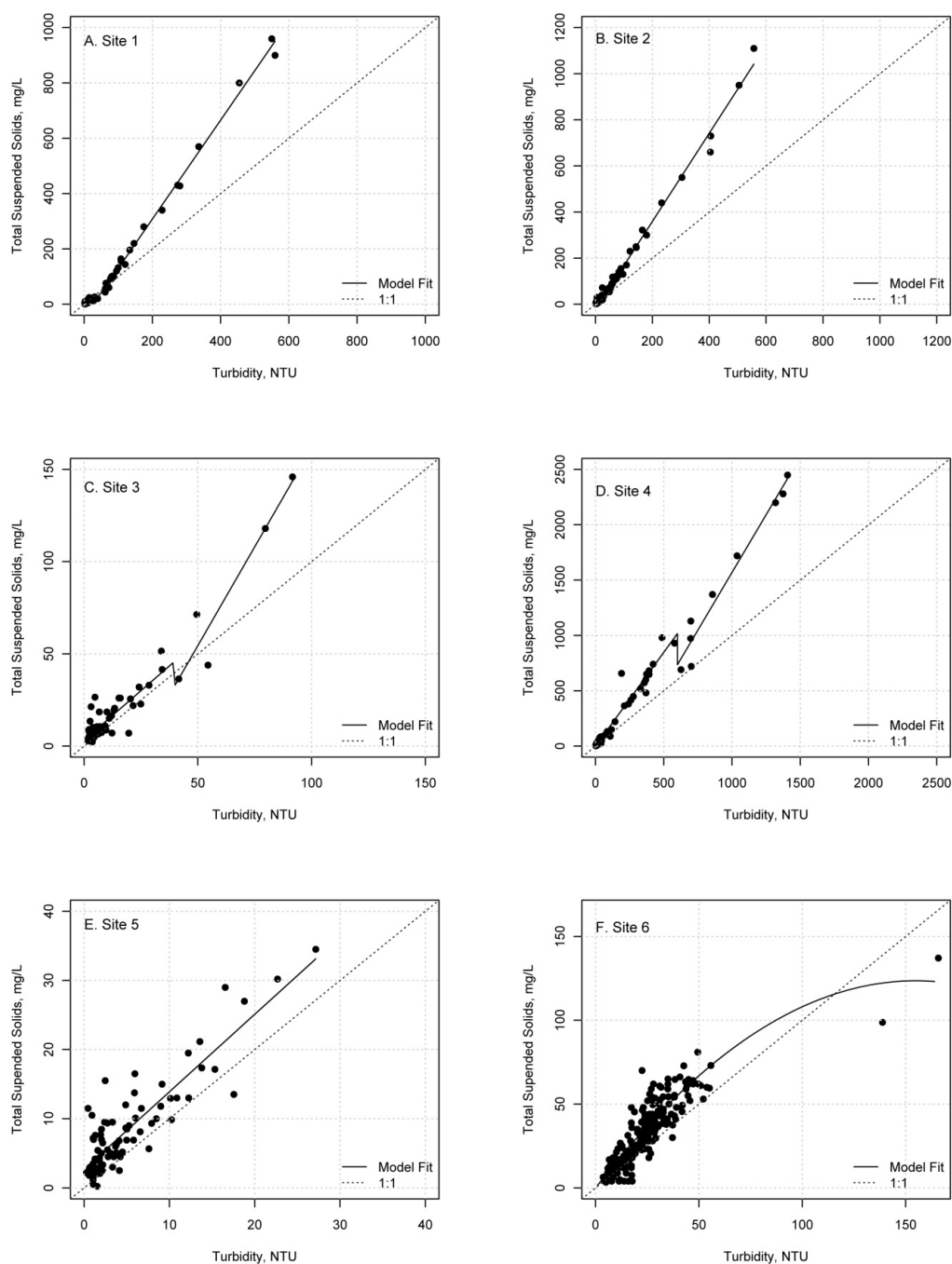Figure 3.2. Scatter plots of total suspended solids (TSS) concentration and turbidity for each of the six monitoring sites. The diagonal dashed line represents what would be a 1:1 relationship between turbidity and total suspended solids (TSS) and is included as a reference against which the steepness of the slopes among sites can be compared. The solid line represents the regression computed (model fit) TSS concentrations
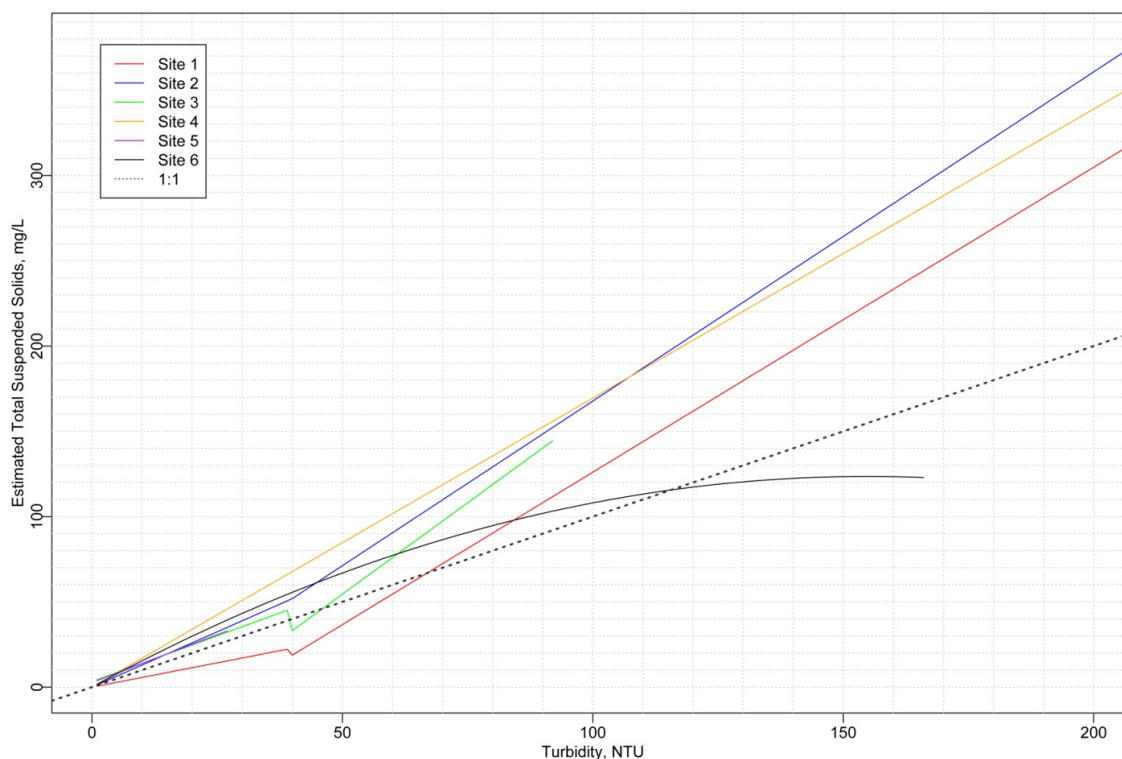
Figure 3.3. Model fit regression lines for each of the six sites in the Little Bear River. The grey dashed line represents what would be a 1:1 relationship between turbidity and total suspended solids (TSS) and is included as a reference against which the steepness of the slopes among sites can be compared. Each of the regression lines represents the estimated TSS concentration (up to 200 NTU) at their respecitve loctions in the watershed (Figure 1). The range of turbidity for which each of the regressions are valid varies among sites and is tabulated in Table 3.3 and illustrated in Figure 3.2
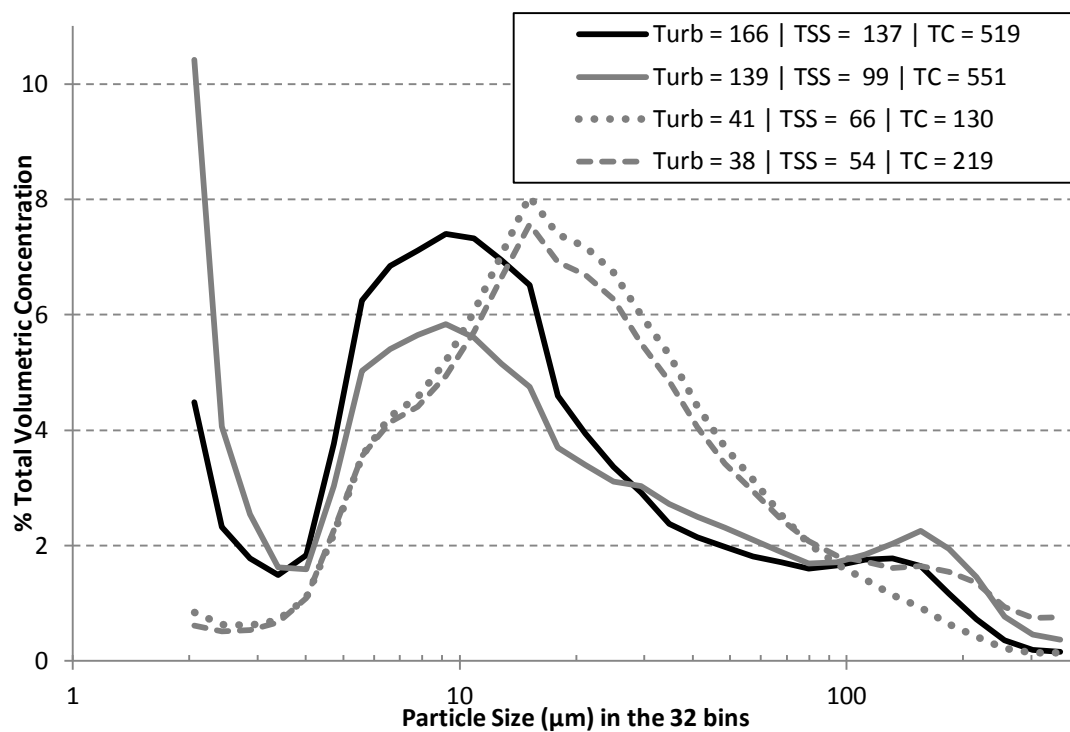
Figure 3.4. Particle size distribution at site 6 for four particle size distributions where Turb=Turbidity (NTU), TSS=total suspended solids (mg/L) and TC=total volumetric concentration (ul/L). The sample dates represent two with high (140-166 NTU) turbidity readings and two with low (38-41) turbidity readings with their respective TSS and total volumetric particle concentrations
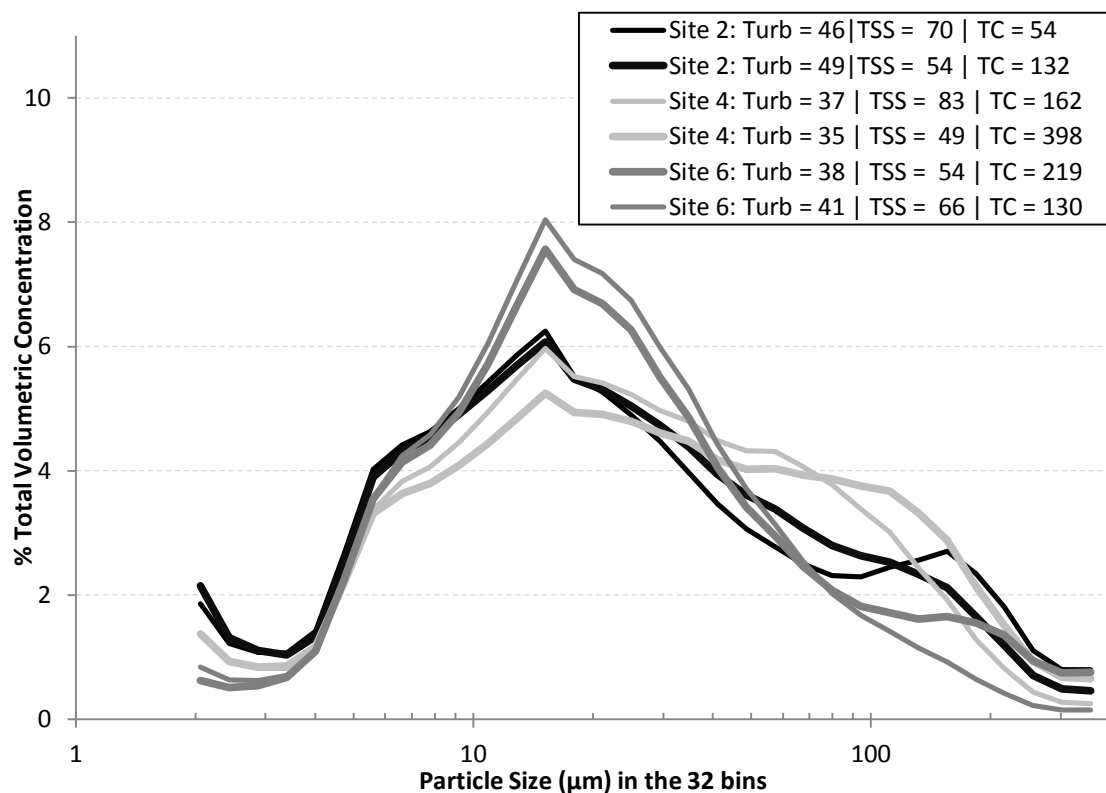
Figure 3.5. Particle size distributions at sites 2, 4 and 6, each having two particle size distributions where Turb=Turbidity (NTU), TSS=total suspended solids (mg/L) and TC=total volumetric concentration (ul/L). The sample dates represent low (35-49 NTU) turbidity readings. Their respective TSS and total volumetric particle concentrations are also given

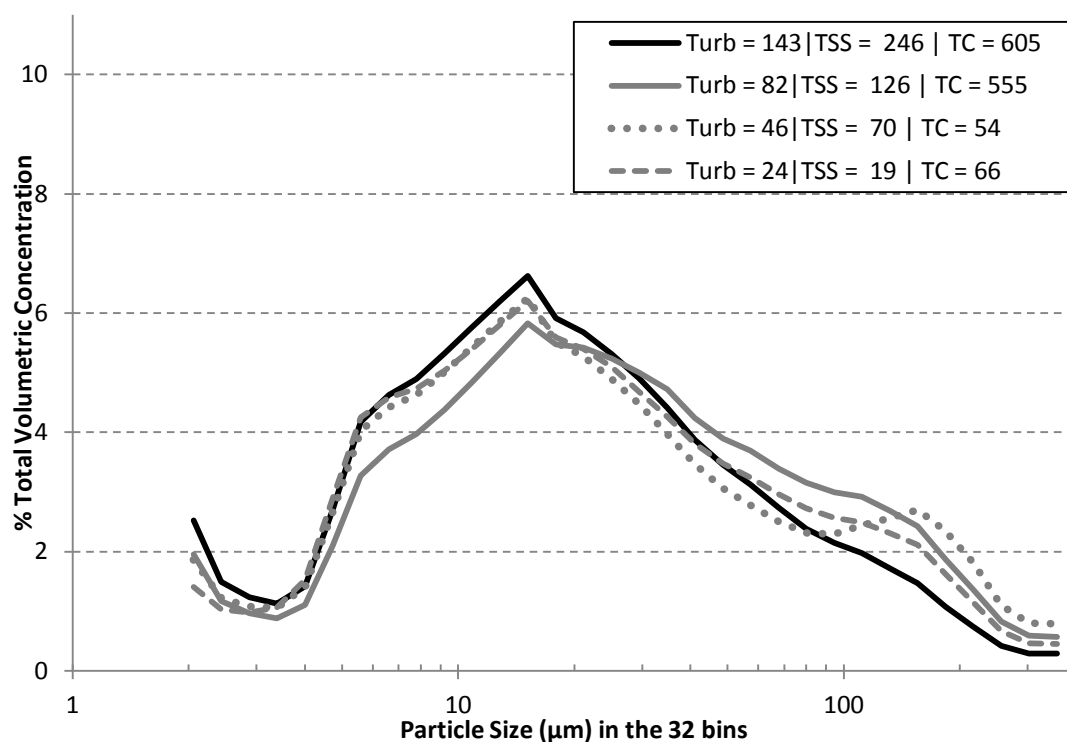Figure 3.6. Particle size distributions for site 2 where Turb=Turbidity (NTU), TSS=total suspended solids (mg/L) and TC=total volumetric concentration (ul/L). Distributions represent both high (82 and 143 NTU) and low (39 and 46 NTU) turbidities

CHAPTER 4

COMPARISON OF POINT-BASED TURBIDITY MEASUREMENTS TO

CROSS SECTION INTEGRATED MEASURES OF SUSPENDED

SOLIDS CONCENTRATION

Abstract

The water quality of streams and rivers is most commonly assessed using single or fixed-point instrument measurements or grab samples that may vary significantly from a cross-section integrated approach. Of particular interest in water quality monitoring is the concentration of suspended sediment, which has many potential negative impacts to a wide variety of beneficial uses. Suspended sediment is most commonly monitored using grab samples, but continuous estimates of suspended sediment derived from turbidity as a surrogate are becoming more common. When using surrogate methods, point-based grab samples of total suspended solids (TSS) concentration are usually correlated with *in situ* turbidity measurements made from a fixed location in the stream channel, and the resulting relationship is used to make high frequency estimates of TSS concentrations. Uncertainty exists, however, in whether grab samples of TSS or turbidity observations collected at a single-point in a stream are representative of the entire steam cross section. The purpose of this study was to investigate differences between point-based and cross-section integrated TSS concentrations and turbidity measurements and to determine whether a method for accounting for these differences in surrogate relationships between TSS and turbidity is needed.

4.1 Introduction

The water quality of streams and rivers is most commonly assessed using single-point measurements or single-point grab samples of water quality variables. In many cases, it is assumed that the stream is a homogeneous mixture of chemical, biological and physical parameters, with similar conditions across the channel cross-section. Of particular interest in water quality monitoring is the concentration and flux of suspended sediment and its associated contaminants. Suspended sediment (including other solid-phase particulates) is cited as the most common impairment to water quality in the United States. Numerous water quality problems are associated with sediment and include potential impacts on nearly all uses of water (AWWA, 1990; Rubin, 1995; Henley et al., 2000; Owens et al., 2001; Soulsby et al., 2001; Pruitt, 2003; Carter et al., 2006; Schoellhamer et al., 2007; Horowitz, 2008; U.S. Army Corps of Engineers, 2011).

The ability to accurately quantify suspended sediment concentrations and fluxes at appropriate temporal and spatial scales is critical to assessing whether streams are meeting their designated beneficial uses and in implementing and evaluating watershed management and mitigation plans. Techniques for quantifying suspended sediment concentration have typically included traditional grab (point-based) sampling methods, which lack sufficient temporal frequency to capture daily, weekly and monthly fluctuations in these concentrations and fluxes (Christensen et al., 2002; Tomlinson and De Carlo, 2003; Kirchner et al., 2004; Horsburgh et al., 2010; Jones et al., 2010). Accurate quantification of suspended sediment concentration and flux at a stream cross

section requires characterizing both the spatial (e.g., across the stream cross section) and temporal variability in suspended sediment concentrations.

One method for quantifying suspended sediment concentrations is to convert high frequency, *in situ* measurements of a surrogate like turbidity into high frequency estimates of suspended sediment concentration using statistical regression techniques that relate *in situ* turbidity measurements and much lower frequency grab samples of suspended sediment (Schoellhamer and Wright, 2003; Rasumussen et al., 2009; Jastram et al., 2010). In most surrogate applications, empirical (surface-grab) sampling methods and *in situ* turbidity measurements are made at a single point location in the stream channel. One potential issue with this is that a single turbidity sensor installed at a monitoring site may not be representative of conditions across the entire stream channel. It can be prohibitively expensive to have multiple turbidity sensors at any one site and logistically impossible to install a sensor in a way that it moves across a stream profile and up and down in the water column.

Others have shown that a single-point measurement of turbidity and suspended sediment may vary significantly from measurements made using width and depth integrated sampling methods (Martin et al., 1992; Horowitz, 2008; Jastram, 2010). Variability in cross-sectional suspended sediment concentrations is typically due to differences in water velocities within the stream channel (also influencing particle settling velocities), incomplete mixing of groundwater inflow, point sources, and upstream tributaries and channel geometry. In some cases, however, it may be possible to develop a correction factor to account for this variability in surrogate regression

equations developed for a particular site. The objective of this study was to investigate the degree to which differences between point and cross-sectionally integrated measurements of turbidity and total suspended solids (TSS) concentration were significant at two water quality monitoring sites in the Little Bear River watershed, Utah, U.S.A. and to investigate methods for mitigating the effect of this potential variability on estimated concentrations and fluxes.

Since turbidity-TSS relationships have already been developed at multiple locations in the Little Bear River watershed (Chapter 3), this study continues efforts to identify factors that can improve the surrogate method for estimating concentrations of suspended sediment using fixed-point turbidity measurements. Data from a single turbidity sensor along with point-based and depth-and-width-integrated grab samples of TSS were compared to assess the spatial distribution of turbidity response and suspended sediment concentration. At sites where a single TSS concentration measurement was not representative of the conditions across the entire stream channel, a correction factor was developed and used to compensate for the difference between a single-point estimate of TSS concentration and an integrated cross-section measurement.

4.2 Study Area

The Little Bear River watershed is located in the south end of Cache Valley, Utah and encompasses roughly 740 km^2. The headwaters form in the Bear River Mountain Range, and the river drains to Cutler Reservoir on the west side of Cache Valley. Elevations within the watershed range from 2865 m to 1345 m. Land use within the upper watershed is comprised mostly of forest and range (grazing) lands while the lower

portion of the watershed is made up primarily of agricultural lands and some urban development. Hydrology of the watershed is dominated by spring snowmelt, with water in the upper watershed being stored within two reservoirs, Porcupine Reservoir in the upper East Fork and Hyrum Reservoir on the mainstem, before being released and diverted into irrigation canals for agricultural use during the irrigation season.

Since 2005, research conducted by the Utah Water Research Laboratory at Utah State University has facilitated the installation of multiple continuous water quality monitoring stations. Two of these sites were used in this study, the locations of which are shown in Figure 1 with descriptions in Table 4.1. One of these stations (site 1) is located in the upper watershed above Hyrum reservoir, where stream channels are steeper and sometimes have velocities as high as 2.0 m/s. The lower portion of the watershed includes the second monitoring site at the outlet of the watershed near the confluence with Cutler Reservoir. In this lower section of the river, channel slopes are lower and velocities are almost always <0.7 m/s. The river exhibits a more meandering planform, and soils are predominantly fine grained lacustrine sediments.

The two sites were selected because accessibility to perform width and depth integrated measurements was facilitated by low-lying bridges that allowed for the specified procedures to be carried out under a full range of flow conditions. Also, these two sites represent hydrologic and water quality conditions that are of interest because they are influenced by distinct watershed characteristics described above.

4.3 Methods

4.3.1 Instrumentation and Data Acquisition

Turbidity, water temperature and gage height were measured using permanently installed, *in situ* sensors every 30 minutes in the Little Bear River. Turbidity and water temperature were measured using a Forest Technology Systems DTS-12 turbidity sensor. The DTS-12 uses a laser diode with a wavelength in the near-infrared (780 nm) spectrum and measures light scatter at 90 degrees to the incident beam (Forest Technologies Ltd, 2010). This geometry results in measurement units of Nephelometric Turbidity Units (NTU), and the probe has a range from 0-1600 NTU. Accuracy of the probe is ± 2% for turbidity values between 0-399 NTU and ±4% for values of between 400-1600 NTU (Forest Technology Systems Ltd., 2007). Each turbidity measurement consists of 100 instantaneous observations made over a five second time interval, after which summary statistics are output. From the 100 instantaneous observations, the median turbidity value was used throughout this study. Water temperature was measured with the DTS-12 from an internal encapsulated thermistor and used to compensate turbidity measurements. Gage height was measured using a KWK Technologies SPXD-600 Pressure Transducer and was correlated with routine discharge measurements to develop a stage-discharge relationship for each site.

4.3.2 Point/Fixed Location Measurements

Point measures of turbidity were made using the above described turbidity sensor at the monitoring sites, with the sensors installed as close as physically possible to being in the main flow of the stream channel (Figures 4.2 and 4.3). For site 1, the sensor is

installed inside a vertical PVC housing that is mounted to a bridge abutment on the up-stream side of the river as shown in Figure 4.2. A rugged metal screen is attached at the submerged end of the housing to protect the protruded sensor face from debris. At site 2, the sensor is also deployed in PVC housing, but its location is in the middle of the up-stream side of the channel on the center concrete abutment of a dual culvert-spanned bridge (Figure 4.3). Both sensors sit ~0.43 m from the bottom of the stream bed. Surface-grab samples for TSS concentration were collected at both sites adjacent to the existing fixed-sensors by either lowering an open wide-mouth bottle approximately 0.3 m below the surface of the water or using a bucket on a rope to retrieve a volume of water for subsequent filling of sample bottles. The bucket on a rope was used at site 2 to collect grab samples of TSS because the bridge is too high above the stream to reach by hand.

4.3.3 Cross Section Measurements

The stream channel cross-sections that were used to perform the width and depth integrated measurements for TSS and turbidity were inline with either the existing fixed turbidity sensor or at the established cross-section where discharge measurements were made. For site 1, the width and depth integrated samples and discharge measurements were taken along the same cross-section, which is ~2 m downstream of the fixed-location turbidity sensor. At site 2, the width and depth integrated samples were collected along the same transect as the fixed turbidity sensor, with discharge measured at a different cross section on the downstream side of the bridge, ~9 m away.

Width and depth integrated TSS measurements followed the equal-width-increment (EWI) method described in Edwards and Glysson (1999). This method specifies that a depth-integrated TSS sample or turbidity measurement should be collected from a minimum of ten equally spaced positions (verticals) along the cross section for streams greater than 1.5 m wide. TSS samples (or turbidity measurements) from each vertical represent the average over the entire depth of the vertical, which spans from the water surface to within 0.09 m of the streambed. The EWI method was preferred over other methods, such as the equal-discharge-increment method (samples taken from centroids of equal-discharge increments), because knowledge of the cross-sectional flow distribution was not required in the field. Because a greater number of verticals are required with the EWI method, variability in the distribution of sediment concentration (and turbidity response) across the cross section could be better defined.

For each site, the wetted width of the stream was determined by spanning a fiberglass tape across the respective cross section. This width was divided into at least ten equally spaced verticals. Using a graduated steel strut, the depth at the horizontal midpoint of each vertical was measured to the nearest 0.003 m. Area near the water's edge where the water depth was less than ~0.15 m or where water velocity was <0.3 m/s or negative (due to stream eddy), was not sampled due to requirements of the suspended sediment sampler. These unsampled areas, however, were included in the gross cross-sectional area calculation.

Prior to making width and depth integrated TSS measurements, stream discharge was determined using a StreamPro acoustic doppler current profiler (ADCP) (Teledyne

RD Instruments Inc., 2006), which measures both water depth and velocities. Maximum depth and velocity were noted for each vertical and were used to calculate the transit rate for the suspended sediment sampler.

A DH-48, depth-integrated, isokinetic (90-110% inflow efficiency) suspended sediment sampler was used for depth-integrated suspended sediment sampling. The DH-48 sampler collects flow-weighted samples and is designed for streams with velocities from 0.3-2.7 m/s and depths up to 2.7 m. The DH-48 samples to within ~0.09 m of the stream bed. The sampler was used at both sites using the manufacturer-supplied extension rods, which allowed it to be lowered and raised from a bridge in the sampling vertical at a specified speed called the transit rate. The transit rate was the same for each vertical and even though the resulting sample volumes from each vertical varied, they were proportional to the discharge of that vertical, resulting in a discharge-weighted suspended sediment sample. The transit rate was determined by the time it takes to traverse the distance from the water surface to the streambed and back in the vertical of maximum discharge that produces a sample volume between 300 and 420 mL.

Depth-and-width-integrated turbidity measurements were made using a DTS-12 turbidity sensor. The sensor was fastened to the graduated steel strut so that it could be lowered from a bridge to the desired depth in each vertical across the stream channel. The sensor was lowered to no less than 0.15 m above the channel bottom with the optical face of the sensor oriented upward and facing upstream. This was done to avoid contact with and potential interferences from stream bed material. Once the sensor was lowered

to the specified depth, measurements were made by the sensor via commands from a laptop computer.

It was not possible to operate the turbidity sensor and the DH-48 at the same time in the same vertical. The internal programming and operation of the DTS-12 sensor requires a single measurement to last ~35seconds, making it impossible to make measurements frequent enough that a continual transit rate for the turbidity sensor in each vertical was feasible. Because of this, turbidity measurements within a vertical followed the sediment sampling, but both were completed within approximately one minute of each other for a given vertical. Additionally, test runs in which turbidity measurements were made at many vertical positions within several verticals showed that turbidity measurements made at 60% of the vertical depth for waters <0.61 m or turbidity measurements made at 20% and 80% of the vertical depth for waters >0.61 m, were sufficient to capture the variability of turbidity within a single vertical. This is similar to the area-velocity method used to compute stream discharge (Chapra, 1997), which requires measurements of velocity at similar intervals within verticals to capture the average velocity representative of the vertical. Since the DH-48 sediment sampler represents an integrated or composite sample for a single vertical, the turbidity measurements were made to reflect the average turbidity of that vertical.

Data collection for a single width and depth integrated TSS and turbidity measurement for a cross section was completed within 30 minutes. To ensure that no significant differences existed between the mobile and fixed turbidity sensors, turbidity measurements made with the mobile sensor were compared to measurements made by the

fixed-point sensor by placing the mobile sensor in near proximity to the fixed-sensor and making measurements over the time periods that the fixed sensor is programmed to operate. This was done for each sampling event before and after width and depth integrated TSS and turbidity measurements were made.

Total suspended solids (TSS) grab samples and depth-integrated samples were analyzed at the Environmental Quality Lab at the Utah Water Research Laboratory using Standard Method 2540D, Total Suspended Solids Dried at 103-105° C (APHA, 1995). Sample bottles collected from each of the verticals were individually analyzed for TSS concentration. Laboratory duplicates for TSS samples were performed on samples collected from the fixed-sensor location and on multiple verticals for a given sampling date.

4.3.4 Comparison of Discrete and Integrated Measurements

Each recorded turbidity measurement is derived from a sample of 100 instantaneous observations. The resulting median and variance (calculated by the sensor) were used for conducting hypothesis tests. The hypothesis tests were used to compare the values of the cross-sectionally integrated measurements with the mean of two discrete measurements that bracket the time window over which the integrated measurements occurred. Hypothesis testing was used to determine if there was a significant statistical difference between the discrete and integrated measurements with a given level of significance of $\alpha=0.05$ (5%). The null hypothesis was that the difference between the discrete and integrated turbidity values is zero (H_0: μ_1-μ_2=0). A p-value less than 0.05

would show statistical reason to reject the null hypothesis and conclude that the discrete and integrated measurements are not statistically the same.

Comparison between discrete and integrated TSS samples was based on the percent differences between duplicate samples that were analyzed in the lab. Percent differences were calculated by dividing the difference between the discrete and integrated samples by the cross-section average TSS concentration and then multiplying by 100.

A correction factor for differences between discrete and integrated TSS concentration samples was developed at site 1 using least squares regression techniques described in Section 3.3.2. Site 1 was chosen for the correction factor development because The data used for the regression analysis was obtained from four sampling events in February – April 2012 where paired discrete (single point grab samples) and integrated (cross section averaged) TSS samples were acquired. These values are listed in Table 4.5. The regression was performed using the four paired data values and the resulting regression equation was used to convert half-hourly regression-computed (based on discrete samples) TSS concentrations to cross section averages. The correction factor developed for use in this study applied a two step adjustment to half-hourly regression-computed TSS concentrations at site 1 for the water years (WY) 2008 and 2011. For the first step, the regression equation was applied to all TSS concentrations less than 246 mg/L. For TSS concentrations greater than 246 mg/L, each value was increased by 7% of the given TSS concentration.

Suspended sediment loads were calculated for water years 2008 and 2011 at site 1 using the following equation:

$$Annual\ Load = \sum_{i=1}^{n} Q_i\ C_i f \hspace{4cm} \text{Eqn. 4.1}$$

where annual load is in metric tons, Q_i are the half-hourly computed discharge values, C_i are the half-hourly, regression-computed TSS concentrations, f is a conversion factor used to convert the product of discharge (cubic feet per second) and concentration (mg/L) to metric tons per year and i is an index for each of half-hourly values where $i = 1,2,…$n. Load calculations were based on 17,564 and 17,240 paired discharge and TSS concentration values for water years 2008 and 2011, respectively.

4.4 Results and Discussion

A total of six paired width and depth integrated turbidity and TSS measurements were collected from the two monitoring locations from February 2012 to April 2012, four at site 1 and two at site 2. An additional two width and depth integrated samples of turbidity were collected within the same time period, one at each site. Low snow pack in the watershed reduced peak discharges and limited high flow/turbidity conditions, resulting in relatively few opportunities for sampling of turbidity and TSS at elevated levels. This has limited the application of these results to a range of conditions that does not fully represent the varying conditions at these two sites. Table 4.3 gives a summary of the number of the turbidity measurements made and the number of TSS samples that were collected and analyzed.

4.4.1 Comparison of Turbidity at Sites 1 and 2

Table 4.4 compares the discrete and integrated measurements of turbidity for the sampling events at both sites. The lower (<15 NTU) turbidity values have small differences between paired measurements, but the difference grows with increasing

turbidity. Also listed in Table 4.4 and illustrated in Figures 4.10 and 4.12 is that the

integrated turbidity values for all sample events are higher than the discrete turbidity

values. For site 1, percent differences between the paired turbidity measurements are

between 17% and 43%. Each of the paired measurements at site 1 were found to be

statistically different, with p-values well below the significance threshold of $\alpha=0.05$.

Although the statistical assurance that the two numbers are different is strong, differences

between paired turbidity values is less than 2 NTU when turbidity is 15 NTU or less. In

the context of estimating TSS concentration using turbidity as a surrogate, the difference

is not significant at this scale. This is because turbidity-TSS data from this site have

shown that turbidity values within ±2 NTU of each other can correlate with TSS grab

samples of the same concentration.

The difference between discrete and integrated turbidity measurements may be

more important, however, for the higher (>40 NTU) turbidity values because that is

where the difference between the paired measurements begins to increase. Figure 4.4

illustrates that not only are the cross-section integrated measurements consistently higher,

but the gap between the paired measurements widens as turbidities increase. At the

highest paired turbidity values of 149 and 191 NTU for the discrete and integrated values

respectively, the percent difference is 22%. The range of turbidity values measured

throughout the cross-section for the integrated sample ranged from 167-200 NTU and

does not include the discrete measurement. This is the case for the remaining sample

events at site 1, where there is no overlap of turbidity in the paired measurements. The

variability in higher turbidity measurements is also illustrated in Figures 4.6 and 4.10.

Figure 4.6 shows cross-section spatial distribution plots for each sample event. The sample events show the average turbidity value for each vertical in the cross section as a shaded area based on the scale of the respective turbidity values. The wide range of turbidities across the stream profile for sample event 3/23/2012 is 140-210 NTU as shown in Figure 4.6. Figure 4.10 shows a comparison of the discrete turbidity measurement and each individual vertical for all turbidity sampling events at site 1. This demonstrates the variability between verticals as turbidity increases.

Because there is not overlap of turbidity in the paired measurements, this would indicate that the fixed sensor is biased lower than cross-section integrated measurements of turbidity up to ~150 NTU. The difference between the paired measurements is thought to be related to the fact that the integrated measurements were made at a position 2 m downstream of the discrete sensor location. The difference between the discrete and integrated measurements is not thought to be instrument related because the mobile sensor that was used to make the integrated measurements was also used to make side-by-side turbidity comparisons with the discrete sensor. On average the measured turbidity values between the two sensors was no more than 10% different.

For site 2, percent differences (Table 4.4) between the paired turbidity measurements were much lower compared to site 1, where each difference was <5%. Paralleling this, the p-values are all greater than 0.9, well above the threshold of $\alpha=0.05$, which demonstrates a lack of evidence to reject H_0: $\mu_1-\mu_2=0$. Additionally, side-by-side comparisons with the mobile and fixed turbidity sensors were made during the field campaigns to identify differences, if any, in turbidity sensor response. Differences

between the sensors were less than 2% for turbidities 0-399 NTU which is in line with the accuracy published in the manufacture's specifications. It is believed that turbidity is more consistent across the stream cross section at this site because the channel is wider, deeper and has a lower slope. The lower and more uniform velocities lead to a more homogenous distribution of turbidity across the channel, and the particles are smaller at this site and more likely to remain suspended.

The cross-section spatial distributions of turbidity at site 2 are shown in Figure 4.8. The range of measured turbidities throughout the cross section were within 4 NTU of each other during all sampling events, with little variation throughout the stream cross-section. Turbidity measurements at the discrete point within the cross section were representative of the entire stream cross-section because all differences between the paired samples were less than 2 NTU. Figure 4.12 shows a comparison of the discrete turbidity measurement with each individual vertical for each of the three sampling events.

4.4.2 Comparison of Total Suspended Solids at Sites 1 and 2

Table 4.5 lists the discrete and integrated samples of TSS for the sampling events at sites 1 and 2. The discrete and integrated concentrations and respective standard deviations for TSS are shown as well as the percent differences between paired samples. The "Lab Precision" column lists the error associated with laboratory duplicate samples. The percent differences between discrete and integrated TSS samples for both sites are generally 10% or less. Sampling event 2/16/2012 at site 1 shows a percent difference of 26% for TSS values (2.4 and 3.25 mg/L), but the values of these samples are below method detection limits of 10 mg/L (U. S. EPA, 1999). Each of the percent differences

between paired samples for both sites are less than their respective laboratory duplicate percent differences. This indicates that the difference in TSS concentration between paired samples is within the error of the analytical procedures carried out in the lab. Therefore differences between the TSS concentrations of the paired samples are not large enough to conclude that surface grab samples are not representative of the cross-sectional average at these two sites and in the range of turbidities sampled.

Differences between discrete and integrated TSS concentrations at site 1 did increase with increasing TSS concentration, but not to the same degree as turbidity. This is illustrated in Figure 4.5, where sample event 3/23/2012 shows that the largest percent difference between paired TSS samples was 7% (the TSS concentrations for the discrete and integrated samples were 246 and 264 mg/L respectively).

Figures 4.7 and 4.11 show the spatial distribution of TSS concentration throughout the verticals for each sample event at site 1. Also illustrated in Figures 4.7 and 4.11 is the location in the stream cross-section where the discrete grab sample was taken, with its size relative to the opening of the sample bottle. Sample events 2/16/2012 and 3/8/2012 had very little variation among sampled verticals, with TSS concentration ranges from 2-6 mg/L and 13-18 mg/L, respectively. The range of TSS concentrations for sample event 3/23/2012 was 200-400 mg/L. Although this represents a large difference of ~100% for concentrations throughout the cross-section, the difference between the discrete grab sample (246 mg/L) and the integrated sample (264 mg/L) was 7%. This demonstrates that at site 1 single-point grab samples could, under certain conditions, approximate the cross-section average of suspended sediment concentration

for values of turbidity <150 NTU. Although turbidity values sometimes exceed 1500 NTU at site 1, the turbidity range of 0-150 NTU is important because it comprises 99% of all measured turbidity values. Other sample events at site 1 also demonstrated small differences between paired samples, as shown in Table 4.5.

However, Figures 4.7 and 4.11 also demonstrate that TSS concentrations from a single vertical could overestimate the measured cross-section average by as much as 122 mg/L where the relative percent difference between the vertical with the highest concentration (386 mg/L) and the overall cross section average (264 mg/L) is ~37%. This observed variability within the cross-section highlights the importance of sensor position and of conducting integrated measurements for use in determining compliance with water quality criteria. Although not captured in this study, grab samples at site 1 with TSS concentrations as high as 1000 mg/L have been collected, and it is expected that at higher TSS concentrations the variability in TSS concentration within the cross section would be larger. This increasing variability among sampled verticals for higher TSS values is illustrated in Figures 4.11.

At site 2, two sampling events for TSS concentration were conducted where percent differences between paired samples were <7% (Table 4.5). As with site 1, the spatial distributions for TSS concentration of site 2 (Figures 4.9 and 4.13) showed variability among verticals. The largest difference from sample event 3/8/2012 was ~70% and was heavily influenced by one vertical that had a concentration of ~50 mg/L. However, TSS concentrations did vary significantly within the cross-section, with TSS concentrations ranging from 14-26 mg/L (~46%) during sample event 2/16/2012 and

from 14-50 mg/L (~112%) during sample event 3/8/2012. Despite the variability within the cross section, the discrete (22.48 mg/L) and integrated (24.28 mg/L) TSS concentrations were within 7% of each other (Table 4.5).

The results have shown that differences between paired discrete and integrated TSS samples are small (~7%) for both sites. While this may indicate—that for a specified range of TSS concentrations—a single discrete sample may sometimes approximate the cross-section average, the impact of these differences on the full range of TSS values for determining annual sediment loads is also of interest. At site 1, this impact was estimated by developing the following correction equation, $y^* = 1.0762(x)$, where y^* is the regression-computed TSS concentration adjusted by the correction factor equation and x is the original regression-computed TSS concentration. This correction equation was used to adjust discrete-based regression computed TSS concentrations (<246 mg/L) so they would be more representative of the integrated cross section average.

Figure 4.14 shows the estimated annual TSS loads at site 1 for the water years 2008 and 2011 based on discrete and integrated (using the correction equation) TSS sample concentrations. For WY 2008 the estimated annual TSS load based on discrete and integrated TSS concentrations was 1,828 MT and 1,967 MT, respectively. For WY 2011, the estimated annual load based on discrete and integrated TSS sampling was 9,742 MT and 10,484 MT, respectively. For both years, the difference between the discrete-based and integrated-based loading estimates was ~7.3%.

For WY 2008, the percentage of annual TSS load that was from TSS concentrations greater than 246 mg/L was 31% for the annual TSS load based on the integrated sample. The percentage of the 2011 annual TSS load that was from TSS concentrations greater than 246 mg/L was 21%. These percentages represent a potentially significant portion of the total annual TSS load for which a correction of 7% was applied. It is believed, however, that the difference between discrete and integrated TSS concentrations would likely increase beyond 7% if TSS concentrations rise, and the percentage of total annual TSS load from these higher levels might accumulate more of the total annual TSS load.

4.5 Conclusions

The purpose of this study was to investigate whether discrete, point-based measures of turbidity and TSS concentration are representative of integrated cross-section averages and to develop a correction factor that accounts for potential differences. Comparisons between measurements of discrete and cross-section integrated turbidity and TSS concentrations were made at two locations in the Little Bear River watershed. At site 1, differences in turbidity between discrete and integrated turbidity values increased to 22% as turbidity increased, whereas at site 2 differences in turbidity between the two measures remained <5%. When turbidity is used as a surrogate for estimating TSS concentration, differences in turbidity between the two methods may be less important because the parameter of concern is TSS concentration. Correcting for differences between discrete measures and integrated measures of turbidity was not necessary for TSS estimation in this study.

Both monitoring sites demonstrated that for some conditions, a single-point grab sample of TSS can be sufficient to represent the cross-sectional average TSS concentration even though many of the sampling periods (Figure 4.7, sample event 3/23/2012; Figure 4.9, sample events 2/16/2012 and 3/8/2012) showed a high degree variability among verticals. For site 1 small differences (~7%) between discrete and integrated TSS samples were found for high (246 mg/L) levels of TSS. This level of TSS concentration is significant because it represents the 99th percentile of all observed turbidities. Because of the high spatial variability among verticals, the location of the sensors in the cross section was a key element to the resulting small differences between discrete and integrated TSS samples. If a sensor had been located in a vertical where differences between discrete and integrated samples were large, then discrete samples of TSS could severely misrepresent the cross section average. A correction factor was developed and used to convert discrete TSS concentrations to TSS concentrations that more closely represent integrated cross section averages.

Total annual TSS loads were estimated using the corrected TSS concentrations and the small differences (~7.3%) between annual TSS loads for both water years does not produce strong reason to use a correction factor for lower (<246 mg/L) TSS concentrations. Above this concentration, however, these results show that a correction factor may be most useful for the higher (>246 mg/L) TSS values because a significant portion (21-31%) of the annual TSS load is derived from these higher TSS concentrations. A correction factor for these higher TSS concentrations may be useful to improve quantification of TSS loads in the Little Bear River.

The implications of these results are applicable to water managers by providing insight to answering key questions like: Where along the stream channel cross section should one retrieve a discrete grab sample for suspended sediment determination that can represent the cross section average? How accurate is a discrete measure of suspended sediment concentration compared to the integrated cross section average? And how can one account for differences between the two measurements? The results show that the methodologies followed in this study can provide direction to answering these questions. These answers can aid water managers in determining how suspended sediment fluxes impact a streams beneficial use.

References

American Public Health Association (APHA). (1995). "Standard methods for the examination of water and wastewater." American Public Health Association, American Water Works Association, and Water Pollution Control Federation, Washington, DC.

American Water Works Association (AWWA). (1990). *Water quality and treatment: A handbook of community water supplies*. McGraw-Hill, New York.

Carter, J., Walling, D. E., Owens, P. N., and Leeks, G. J. L. (2006). "Spatial and temporal variability in the concentration and speciation of metals in suspended sediment transported by the River Aire, Yorkshire, UK." *Hydrological Processes*, 20(14), 3007-3027.

Chapra, S. C. (1997). *Surface water-quality modeling*. McGraw-Hill, New York.

Christensen, V., Rasmussen, P., and Ziegler, A. (2002). "Real-time water quality monitoring and regression analysis to estimate nutrient and bacteria concentrations in Kansas streams." *Water Science and Technology*, <http://ks.water.usgs.gov/pubs/reports/vgc.0610.html> (July 15, 2011).

Edwards, T. K., and Glysson, G. D. (1999). "Field methods for measurement of fluvial sediment, U.S. Geological Survey Techniques of Water-Resources Investigations, book 3, chap. c2." <http://pubs.water.usgs.gov/twri3c2> (October 4, 2011).

Forest Technology Systems, Ltd. (2007). DTS-12 SDI-12 turbidity sensor operating manual. Forest Technology Systems, Victoria, BC.

Henley, W. F., Patterson, M. A., Neves, R. J., and Lemly, A. D. (2000). "Effects of sedimentation and turbidity on lotic food webs: A concise review for natural resource managers." *Reviews in Fisheries Science*, (August 2011), 37-41.

Horowitz, A. J. (2008). "Determining annual suspended sediment and sediment-associated trace element and nutrient fluxes." *The Science of the Total Environment*, Elsevier B.V., 400(1-3), 315-43.

Horsburgh, J. S., Spackman Jones, A., Stevens, D. K., Tarboton, D. G., and Mesner, N. O. (2010). "A sensor network for high frequency estimation of water quality constituent fluxes using surrogates." *Environmental Modeling & Software*, Elsevier Ltd, 25(9), 1031-1044.

Jastram, J. D., Zipper, C. E., Zelazny, L. W., and Hyer, K. E. (2010). "Increasing precision of turbidity-based suspended sediment concentration and load Estimates." *Journal of Environmental Quality*, 1306.

Jones, A. S., Stevens, D. K., Horsburgh, J. S., and Mesner, N. O. (2010). "Surrogate measures for providing high frequency estimates of total suspended solids and total phosphorus concentrations." *Journal of the American Water Resources Association*, 47(2), 239-253.

Kirchner, J. W., Feng, X., Neal, C., and Robson, A. J. (2004). "The fine structure of water-quality dynamics: the(high-frequency) wave of the future." *Hydrological Processes*, 18(7), 1353-1359.

Martin, G., Smoot, J., and White, K. D. (1992). "A comparison of surface-grab and cross sectionally integrated stream-water-quality sampling methods." *Water Environment Research*, 64(7), 866-876.

Owens, P. N., Walling, D. E., Carton, J., Meharg, A. A., Wright, J., and Leeks, G. J. L. (2001). "Downstream changes in the transport and storage of sediment-associated contaminants (P, Cr and PCBs) in agricultural and industrialized drainage basins." *The Science of the Total Environment*, 266(1-3), 177-86.

Pruitt, B. A. (2003). "Uses of turbidity by states and tribes." Proceedings of the Federal Interagency Workshop on Turbidity and Other Sediment Surrogates, U.S Geological Survey, Reno, NV, April 30-May 2, 2002.

Rasmussen, P. P., Gray, J. R., Glysson, G. D., and Ziegler, A. C. (2009). "Guidelines and procedures for computing time-series suspended-sediment concentrations and loads from in-stream turbidity-sensor and streamflow data." U.S. Geological Survey Techniques and Methods book 3, chap. C4, 53 p.

Rubin, J. F. (1995). "Estimating the success of natural spawning of salmonids in streams." *Journal of Fish Biology*, Wiley Online Library, 46(4), 603-622.

Schoellhamer, D.H., and Wright, S. A. (2003). "Continuous measurement of suspended-sediment discharge in rivers by use of optical backscatterance sensors." *IAHS Publication*, (June 2002), 28-36.

Schoellhamer, D. H., Mumley, T. E., and Leatherbarrow, J. E. (2007). "Suspended sediment and sediment-associated contaminants in San Francisco Bay." *Environmental Research*, 105(1), 119-31.

Soulsby, C., Youngson, A. F., Moir, H. J., and Malcolm, I. A. (2001). "Fine sediment influence on salmonid spawning habitat in a lowland agricultural stream: A preliminary assessment." *The Science of the Total Environment*, 265(1-3), 295-307.

Stubblefield, A. P., Reuter, J. E., Dahlgren, R. A., and Goldman, C. R. (2007). "Use of turbidometry to characterize suspended sediment and phosphorus fluxes in the Lake Tahoe basin, California, USA." *Hydrological Processes*, Wiley Online Library, 21(3), 281-291.

Teledyne RD Instruments Inc. (2006). *StreamPro ADCP Steamflow measurement system operating manual*. Teledyne RD Instruments, Ponway, CA, U.S.A.

Tomlinson, M. S., and De Carlo, E. H. (2003). "The need for high resolution time series data to characterize Hawaiian streams." *Journal of the American Water Resources Association*, 39(1), 113-123.

U.S. Army Corps of Engineers. (2011). "Maintenance dredging and the environment." <http://www.nab.usace.army.mil/whatwedo/dredging.htm> (Aug. 1, 2011).

U.S Environmental Protection Agency (U.S. EPA). (1999). "Total suspended solids (TSS) EPA Method 160.2." *EPA*, 2, 1-3. <http://www.epa.gov/region9/qa/pdfs/160_2.pdf > (July 11, 2011).

Table 4.1. Summary of water quality monitoring sites in the Little Bear River watershed. The date range reflects the time period over which continuous monitoring has been conducted

Site Number	Site Name	Date Range	Latitude	Longitude	Site Description
1	Lower South Fork	July 2007-Present	41.5065	-111.8151	Located below the confluence of the Upper South Fork and a major tributary, Davenport creek. Land use is dominantly forest and grazing
2	Mendon	August 2005-Present	41.7185	-111.9464	Located near the outlet of the watershed at Cutler reservoir. Water quality is affected by agriculture return flows and the Wellsville WWTP

Table 4.2. Summary of water quality sensor specifications

Variable	Sensor	Specifications
Turbidity	DTS-12 turbidity sensor, Forest Technology Systems, Inc.	Accuracy: $\pm$ 2% 0-399 NTU and $\pm$ 4% 400-1600 NTU
Water Temperature	Encapsulated thermistor sensor, Forest Technology Systems, Inc.	Accuracy: $\pm 0.2^{\circ}$C
Stage	SPXD-600 pressure transducer, KWK Technologies, Inc.	Accuracy: $\pm$ 1% of the full measurement span

Table 4.3. Summary of the number of turbidity measurements and number of TSS samples collected and analyzed for each sampling event and for each site. Discrete turbidity measurements represent the average of the two median turbidity values (each one based on 100 instantaneous observations) that bracketed the time period over which the integrated measurements were made. Discrete TSS samples are laboratory duplicates

Site	Sampling Event	Discrete Turbidity Measurements	Integrated Turbidity Measurements	Discrete TSS Samples	Integrated TSS Samples
1	2/14/2012	2	20	-	-
	2/16/2012	2	22	2	11
	3/8/2012	2	22	2	9
	3/23/2012	2	19	2	10
	4/26/2012	2	18	2	11
2	2/8/2012	2	22	-	-
	2/16/2012	2	29	2	12
	3/8/2012	2	22	2	12

Table 4.4. Comparison of discrete and integrated turbidity measurements for the two monitoring sites. Percent differences between paired measurements are shown. The p-value indicates a statistical significance that the difference between the paired measurements is not zero. The threshold for significance is α=0.05

Site	Sampling Event	Discrete Turbidity (NTU)	Discrete Std. Dev. (NTU)	Integrated Turbidity (NTU)	Integrated Std. Dev. (NTU)	Diff.	*p*-value
1	2/14/2012	1.9	0.83	2.97	0.25	35%	<<0.001
	2/16/2012	1.5	0.65	2.52	0.13	43%	<<0.001
	3/8/2012	11.5	1.2	13.73	0.74	17%	<<0.001
	3/23/2012	149.4	6.6	191.3	14.5	22%	<<0.001
	4/26/2012	39.8	2.6	48.5	2.8	18%	<<0.001
2	2/8/2012	21.9	2.0	21.7	1.8	<1%	0.93
	2/16/2012	14.3	1.2	13.9	0.95	<3%	1.8
	3/8/2012	17.6	1.6	16.8	1.1	4.7%	1.9

Table 4.5. Comparison of TSS between point-based and cross-sectional integrated samples for the two monitoring sites. Percent differences between paired samples are shown with the percent difference of laboratory duplicates also shown

Site	Sampling Event	Discrete TSS (mg/L)	Discrete Std. Dev. (mg/L)	Integrated TSS (mg/L)	Integrated Std. Dev. (mg/L)	Difference	Lab precision
1	2/16/2012	2.40	0.57	3.3	1.3	26%	33%
	3/8/2012	15.6	1.1	14.2	1.6	10%	10%
	3/23/2012	246	41.1	264	34.8	7%	24%
	4/26/2012	57.1	3.50	61.4	5.67	7%	28%
2	2/16/2012	18.8	1.14	18.5	3.1	1%	9%
	3/8/2012	22.5	0.67	24.3	9.3	7%	4%
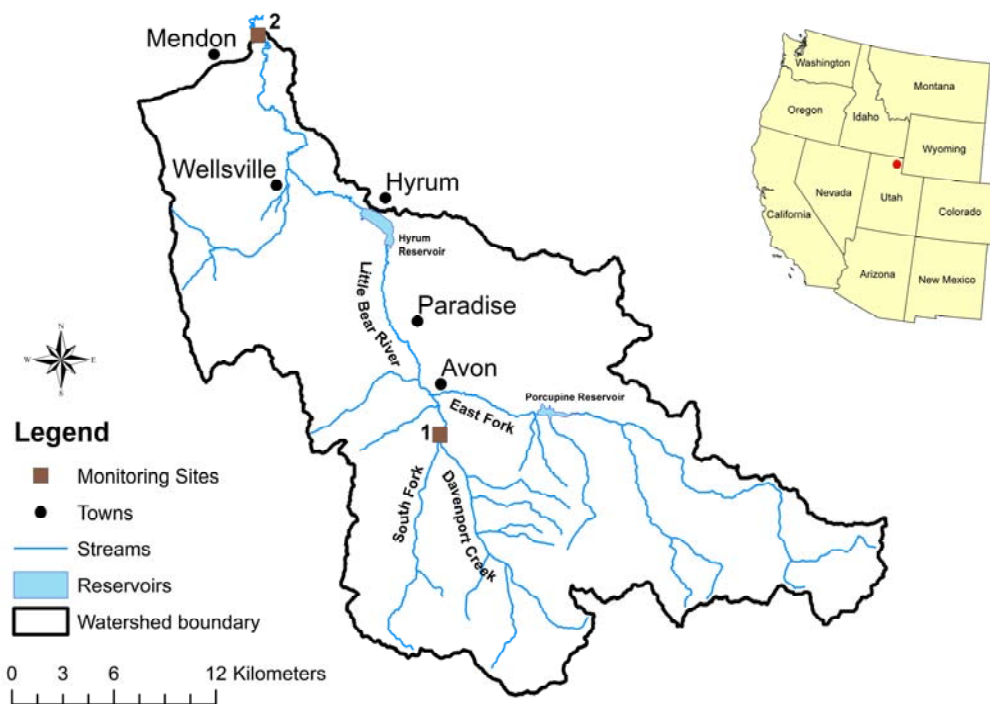
Figure 4.1. Little Bear River watershed water quality monitoring sites 1 and 2

Figure 4.2. Photograph of the fixed-location turbidity sensor relative to dominant stream flow pathway for Site 1. Sensor housings are the black vertical columns located underneath the bridge on the right-hand side of the photograph

Figure 4.3. Photograph of the fixed-location turbidity sensor relative to channel cross-section for Site 2. Sensor housings are the black vertical columns in the upper center of the photograph
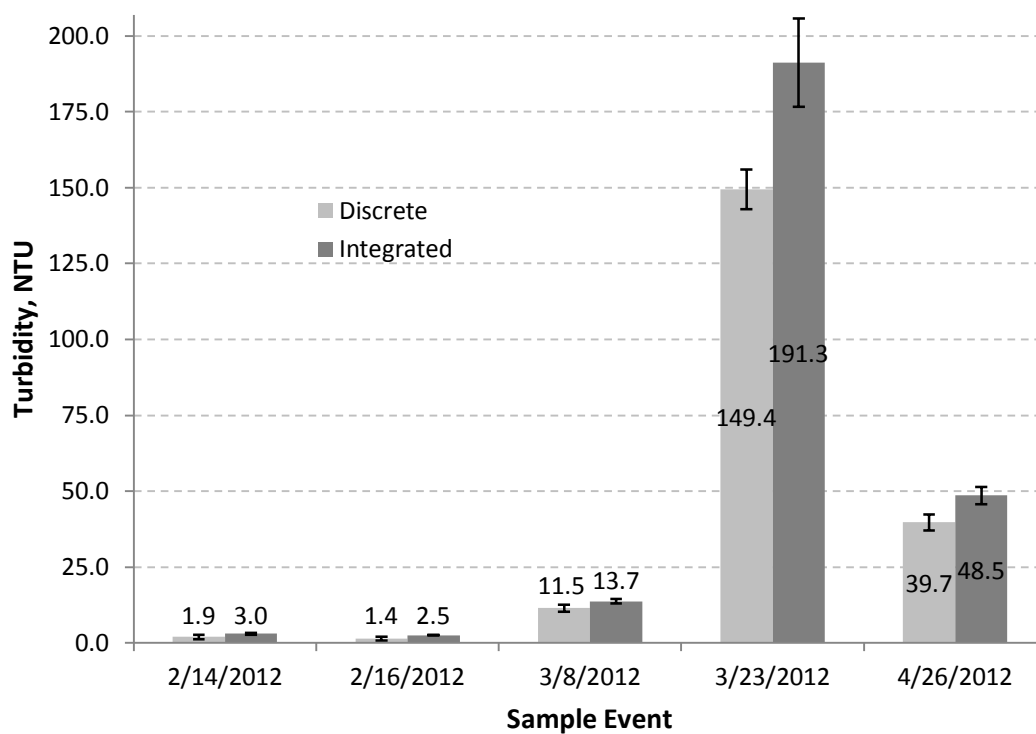
Figure 4.4. Difference between paired discrete and integrated turbidity measurements at site 1. At sample event 3/23/2012 the percent difference is 22%. For the discrete measurements, error bars represent the standard deviation of a turbidity measurement made by the *in situ* sensor. For the integrated turbidity measurements, the error bars represent the standard deviation of the integrated measurements
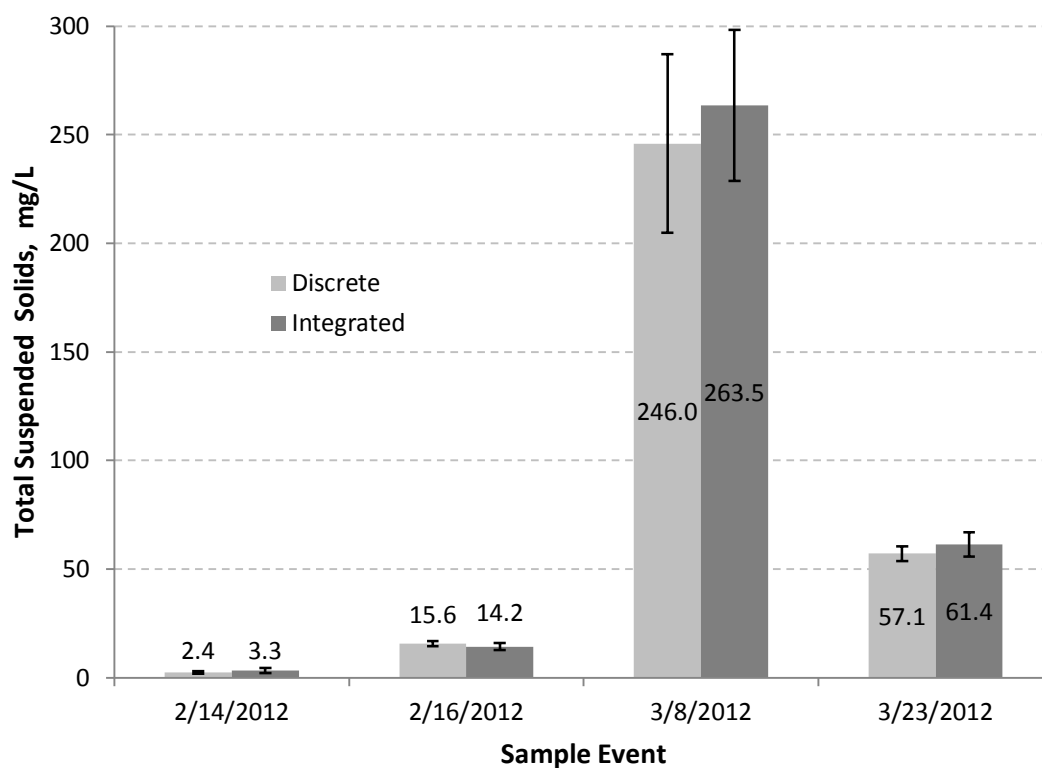
Figure 4.5. Difference between paired discrete and integrated total suspended solids concentrations at site 1. At sample event 03/23/12 the percent difference is 7%. For the discrete measurements, error bars represent the standard deviation of duplicate TSS samples. For the integrated TSS measurements, the error bars represent the standard deviation of the integrated measurements
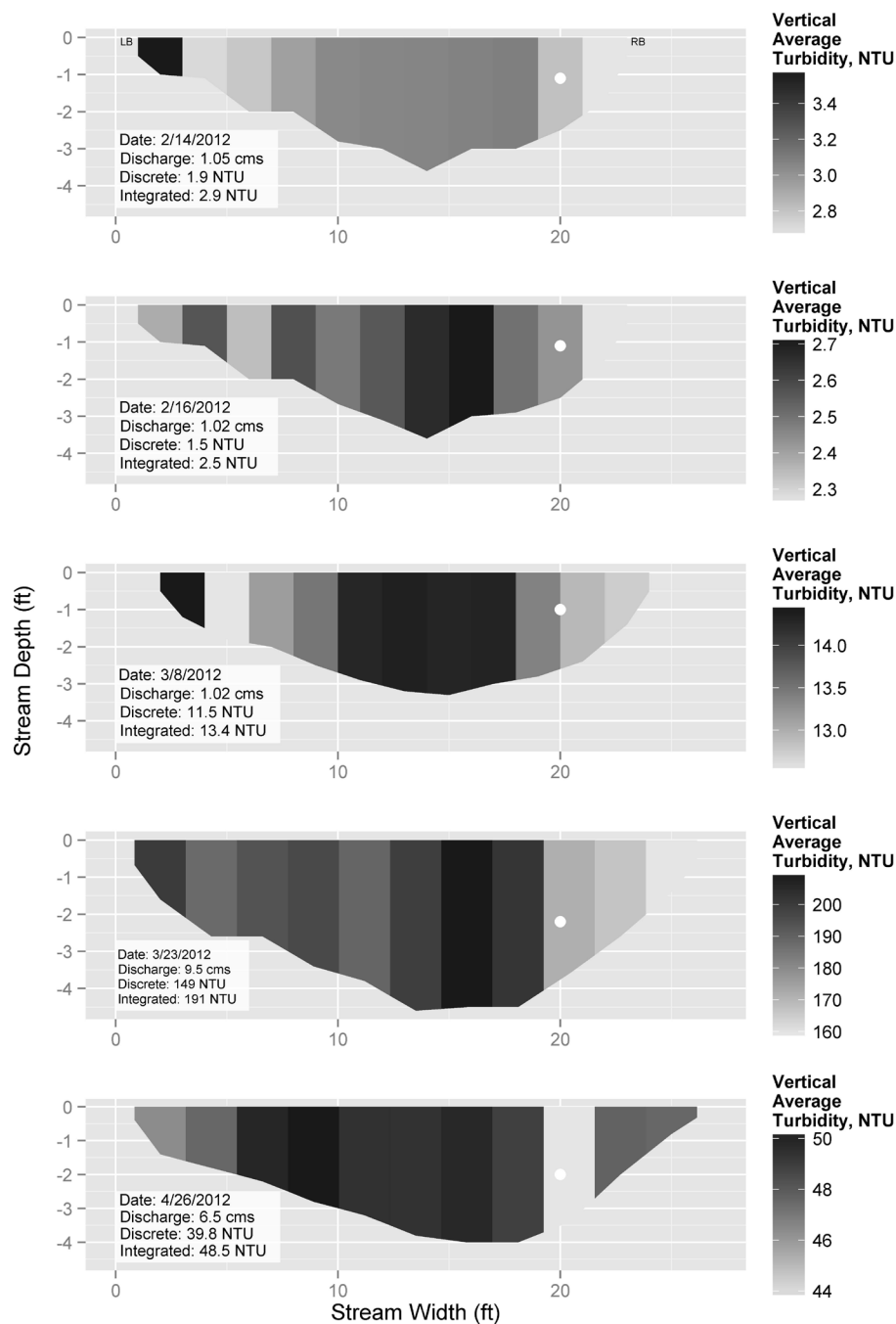
Figure 4.6. Spatial distribution of the average turbidity for each vertical for each of the five sampling events at site 1. The legend scale on the right-hand side represents the range of turbidity values throughout the cross section for the respective sample event. The white circles represent the location of the fixed turbidity sensor and the location at which discrete measurements were made. Its size represents the approximate area measured by the sensor
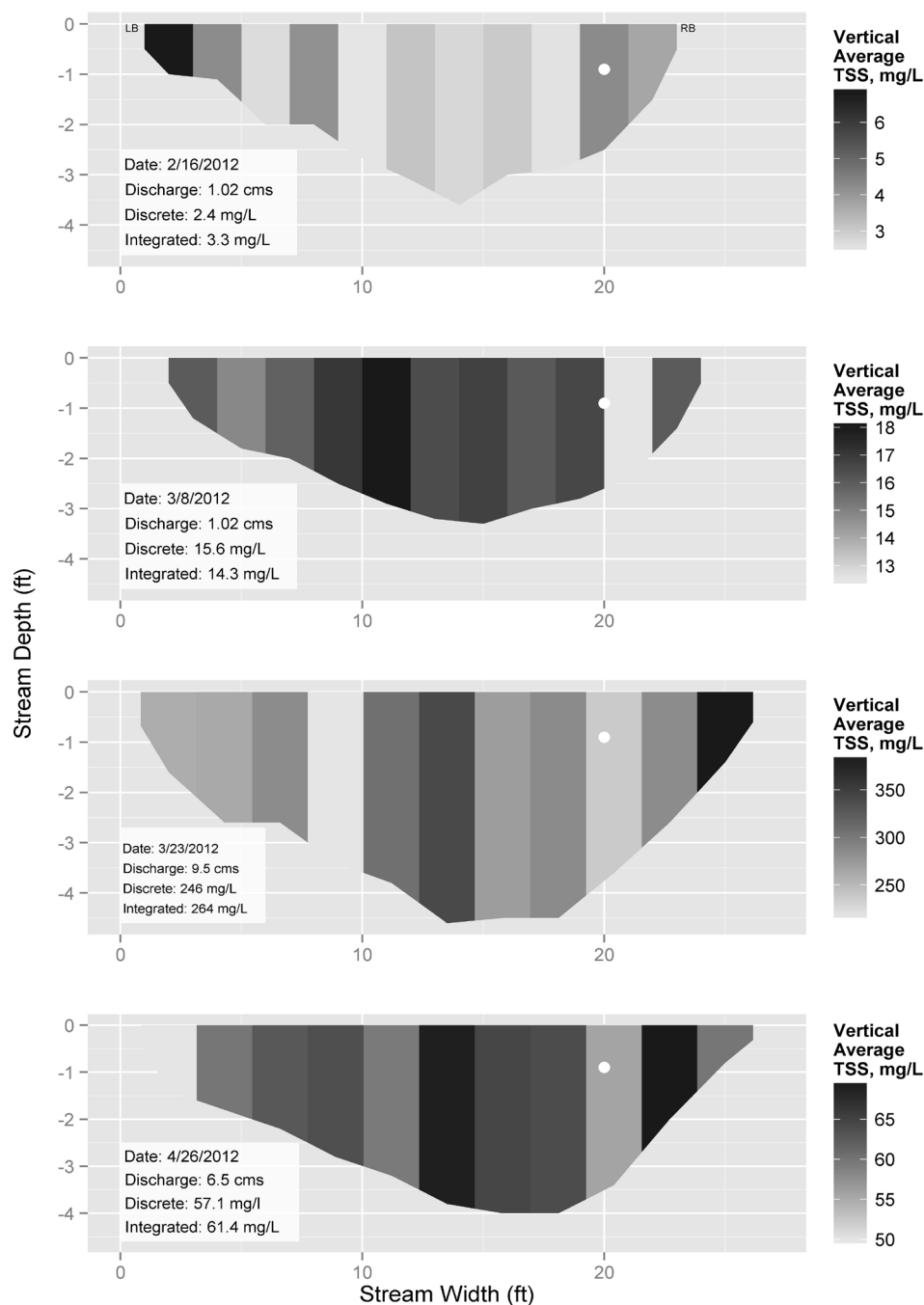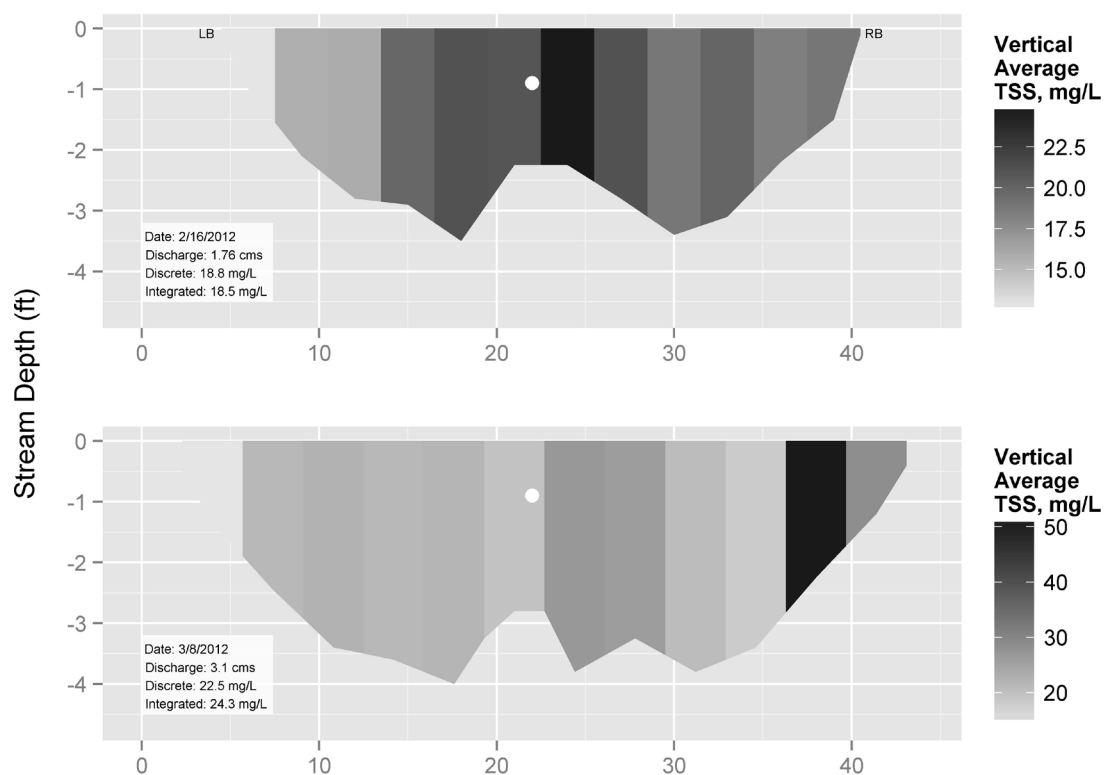
Figure 4.7. Spatial distribution of the average total suspended solids (TSS) concentration for each vertical for each of the four sampling events at site 1. The legend scale on the right-hand side represents the range of TSS values throughout the cross section for the respective sample event. The white circles represent the approximate location where the grab samples were taken and the location. Its size represents the approximate area sampled by the bottle
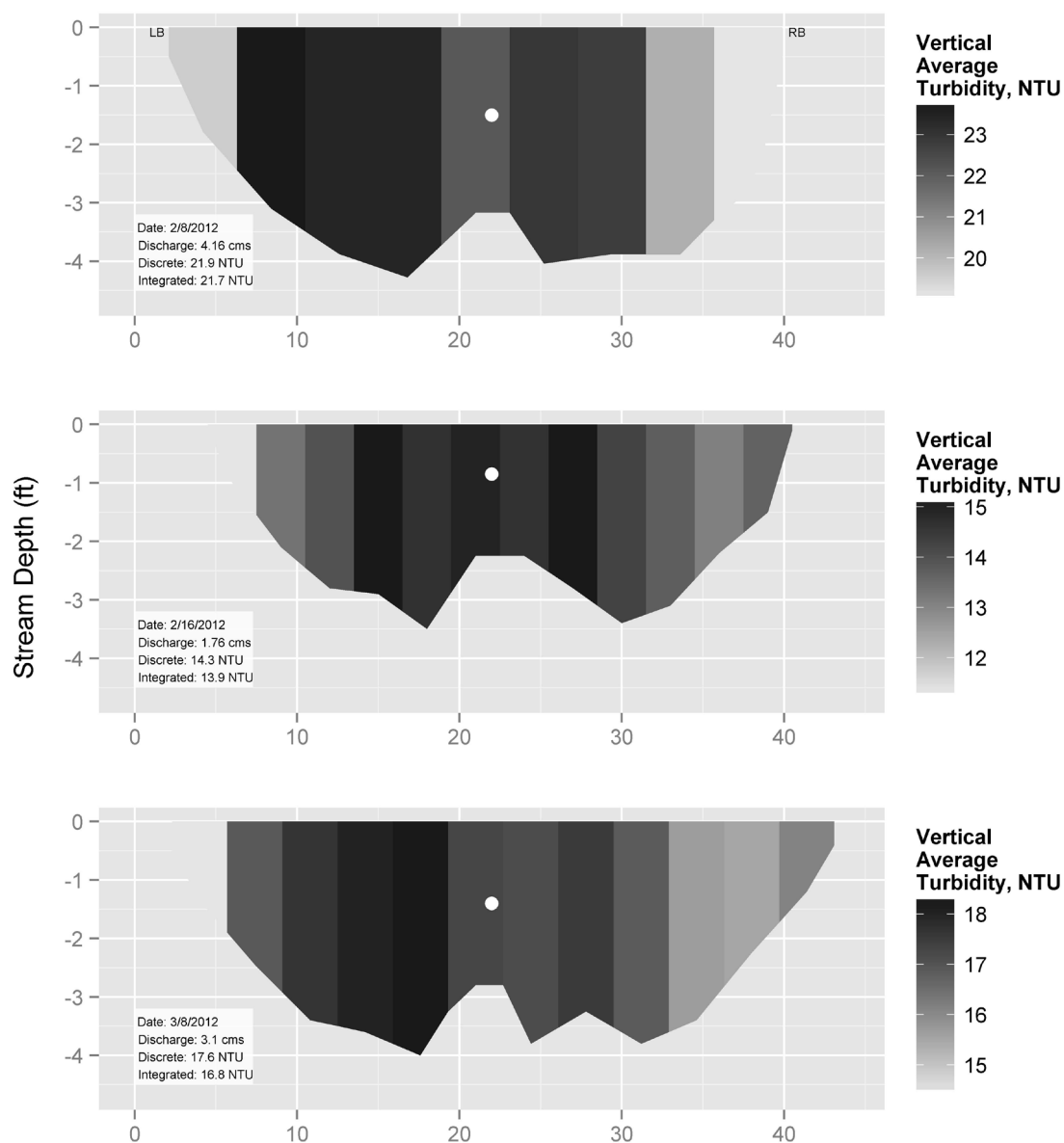
Figure 4.8. Spatial distribution of the average turbidity for each vertical for each of the three sampling events at site 2. The legend scale on the right-hand side represents the range of turbidity values throughout the cross section for the respective sample event. The white circles represent the location of the fixed turbidity sensor and the location at which discrete measurements were made. Its size represents the approximate area measured by the sensor

Figure 4.9. Spatial distribution of the average total suspended solids (TSS) concentration for each vertical for each of the two sampling events at site 2. The legend scale on the right-hand side represents the range of TSS values throughout the cross section for the respective sample event. The white circles represent the approximate location where the grab samples were taken and the location of the sensor. Its size represents the approximate area sampled by the bottle
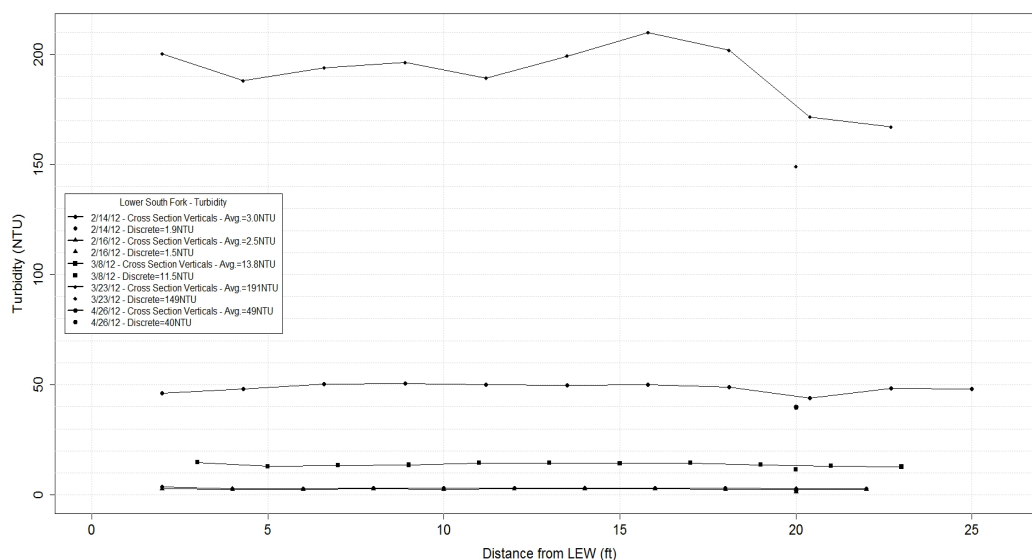
Figure 4.10. Comparison of discrete turbidity measurements to each sampled vertical across the cross section for site 1
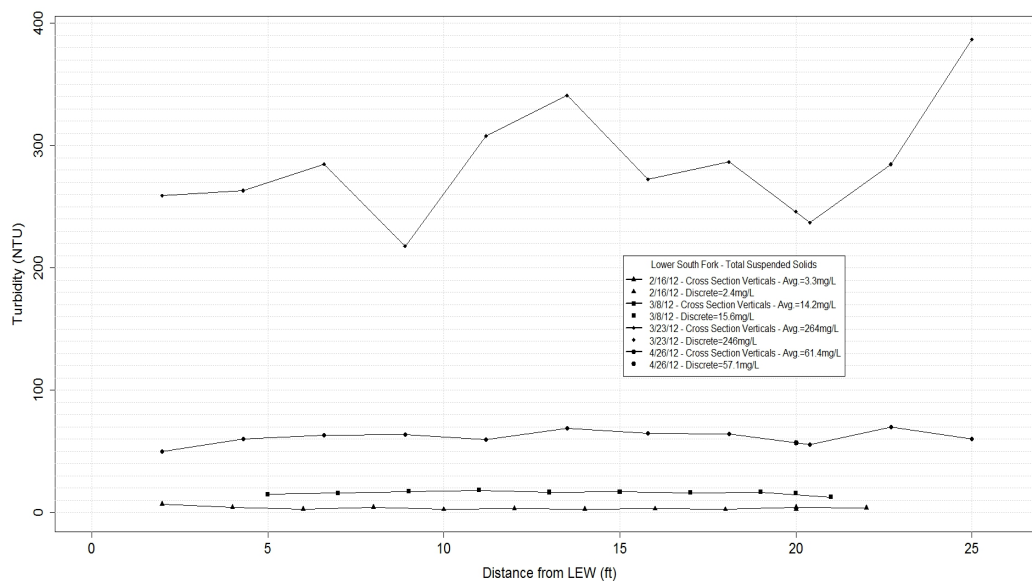
Figure 4.11. Comparison of discrete TSS samples to each sampled vertical across the cross section for site 1
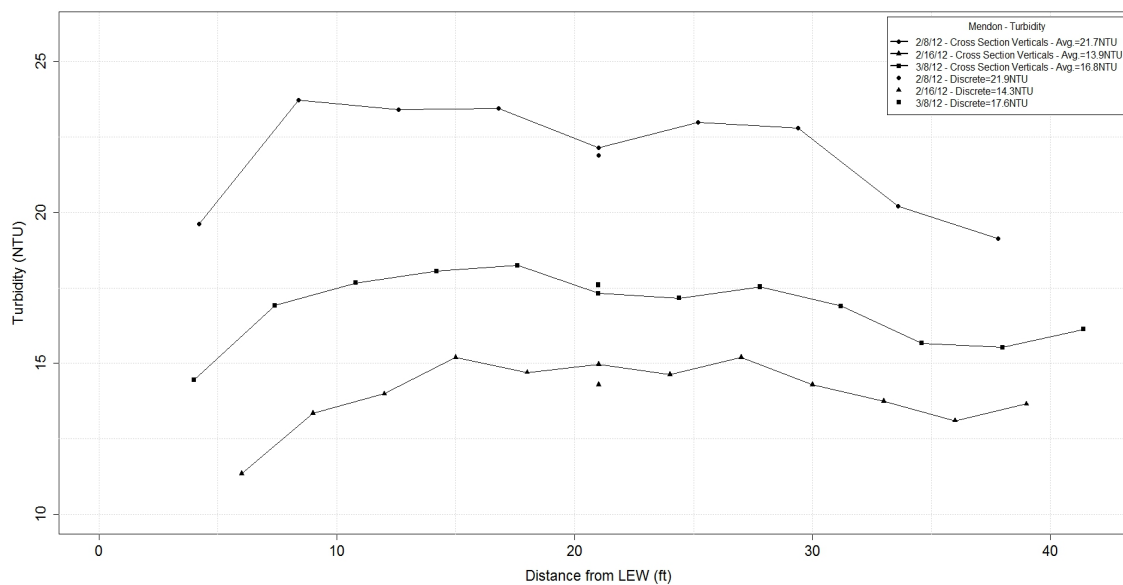
Figure 4.12. Comparison of discrete turbidity measurements to each sampled vertical across the cross section for site 2
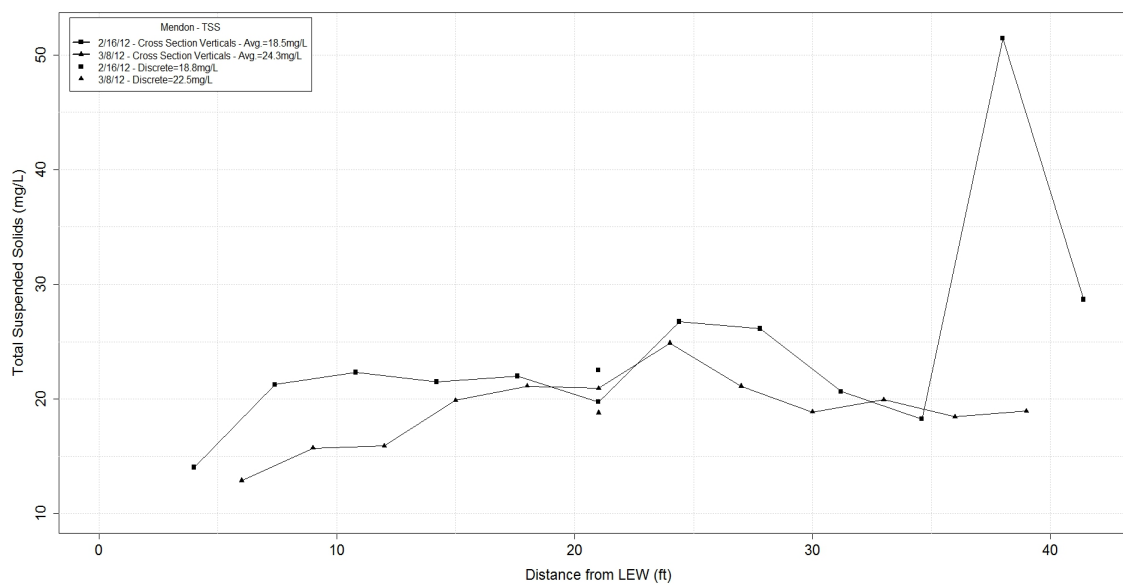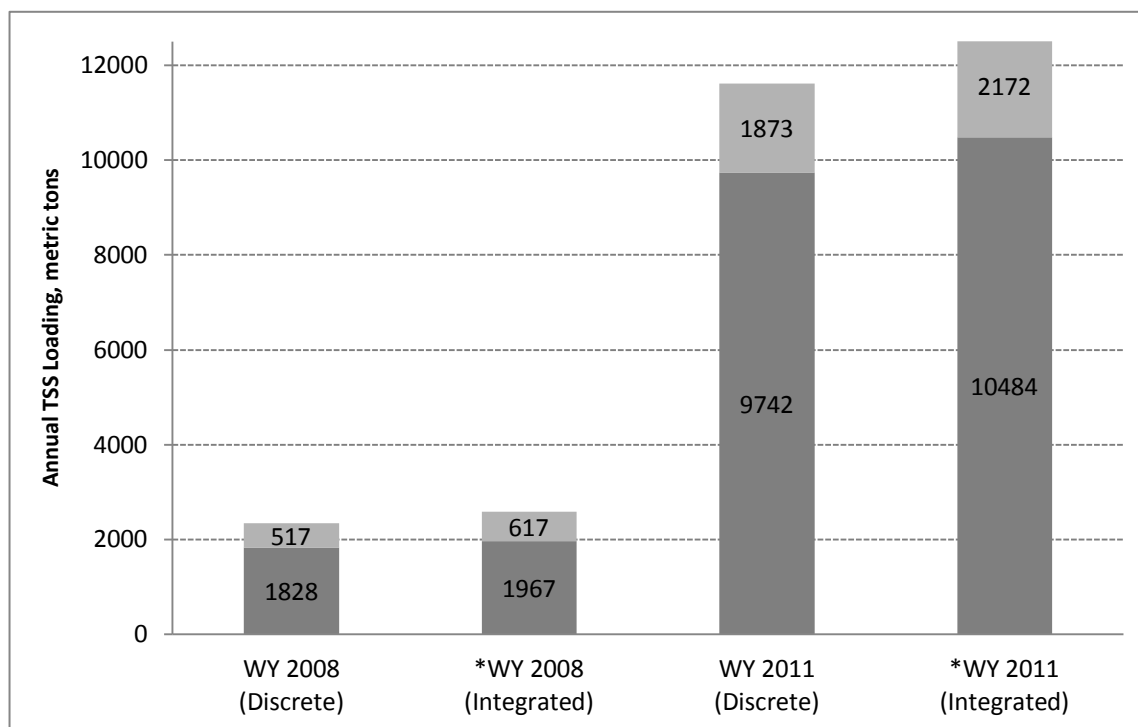

Figure 4.13. Comparison of discrete TSS samples to each sampled vertical across the cross section for site 2

Figure 4.14. Annual total suspended solids (TSS) loading estimates in metric tons for the water years (WY) 2008 and 2011 at site 1. For WY 2008 and 2011 noted with an "*" a correction factor has been applied to the estimated TSS concentrations that were used to calculate the TSS loads. The lighter shaded bars represent the portion of the TSS load that is from TSS concentrations greater than 246 mg/L

CHAPTER 5

ASSESSING UNCERTAINTY IN CONTINUOUS

CONCENTRATIONS OF TOTAL SUSPENDED SOLIDS DERIVED

FROM TURBIDITY

Abstract

Continuous estimates of suspended sediment concentration derived from surrogate measures such as turbidity have the potential to aid in better understanding the timing and magnitude of suspended sediment concentrations and loads and in determining whether streams are attaining or exceeding established water quality criteria. The high temporal resolution of continuous measurements of turbidity is better able to capture the timing and magnitude of concentrations and fluxes than more traditional grab sampling techniques. However, uncertainty in continuous suspended sediment estimates derived from turbidity must be estimated to provide proper context for their interpretation. This study quantified the uncertainty associated with turbidity measurements, analytical measurements of suspended sediment concentration (TSS) and regression computed TSS concentrations at two sites in the Little Bear River, Utah, USA. A regression model was used to compute continuous time series of TSS, and then a 90 % prediction interval that incorporates this uncertainty was calculated and used to assess compliance with a water quality criterion value for suspended sediment that was established during a total maximum daily load (TMDL) study of the Little Bear River. The percent of estimated TSS concentrations exceeding the water quality criterion was

estimated with statistical confidence while accounting for uncertainty introduced from measurement and regression techniques.

5.1 Introduction

A potential use for high frequency estimates of suspended sediment concentrations derived from surrogates such as turbidity using regression techniques is in determining whether streams are attaining or exceeding an established water quality standard or water quality criteria developed for a special purpose such as a total maximum daily load (TMDL). Due to the high temporal resolution that is gained from continuous measurements of turbidity, sediment concentration estimates derived from continuous turbidity measurements can reveal critical information related to the timing and magnitude of concentrations and loads, which may play a key role in assessing compliance with water quality standards. However, uncertainty in the estimates must be quantified to provide the proper context for their interpretation and aid in determining the confidence with which these estimates can be used for assessing compliance with water quality criteria.

A number of authors have computed high frequency estimates of suspended sediment concentration from empirical relationships using turbidity and least squares regression analysis (Gippel, 1995; Minella et al., 2008; Rasmussen et al., 2009; Jastram et al., 2010; Jones et al., 2010). Regression models provide a framework for which the quantification of uncertainty in continuous estimates of suspended sediment concentration can be determined. One approach involves computing prediction intervals that specify a range of values within which a predicted concentration can be expected to

fall. Because the prediction intervals express a range of concentrations within which the actual concentration is expected to fall (rather than a single value), the bounds of the prediction interval can be used to estimate a probable range of percentage values within which the actual percent exceedance of a water quality criterion (WQC) value is expected to fall (rather than a single value).

Prediction intervals, however, have the potential to overstate the confidence in estimated suspended sediment concentrations if the errors in the analytical and regression procedures are not considered. Uncertainty in suspended sediment concentrations estimated using surrogate methods arises from (1) error in the measurements of the regression variables (i.e., turbidity and suspended sediment concentration); and (2) statistical uncertainty in the regression relationship. If unaccounted for, the error with which turbidity and suspended sediment concentration are measured can increase chances for erroneous regression model outcomes (Bertrand-Krajewski, 2006) and increase the uncertainty in determining compliance with water quality standards using regression computed concentration estimates.

Turbidity measurements are affected by instrument drift, fouling and sensor malfunction, each of which can be minimized with maintenance practices. The precision with which turbidity can be measured is quantifiable because instrument technology allows for rapid, consistent measurements that can be repeated for a wide range of turbidity values. Manufacturer specifications often provide information related to sensor accuracy, which can be used to adjust in stream measurements.

Suspended sediment concentration measurements are affected by a variety of procedures that occur in the collection, preservation and storage of samples as well as laboratory materials, handling, equipment and methods. The careful and consistent handling and processing of suspended sediment samples following methods outlined by established analytical procedures can reduce much of the error associated with suspended sediment concentration determination and can yield uncertainties of approximately 25-30 % (Bertrand-Krajewski, 2006). Bertrand-Krajewski (2006), and Rasmussen et al., (2009) cite the necessity of collecting a representative suspended sediment sample that is descriptive of the entire stream cross section. They note that point samples may under or over estimate cross-section averages by as much as 50 % and that this single measure may be the greatest source of uncertainty in developing a turbidity-suspended sediment estimation model.

Quantifying uncertainty in the measurements of water quality parameters such as TSS provides a necessary context to which water managers and regulators may make decisions that have significant implications. For example, language from a TMDL study of the Little Bear River in northern Utah, USA states that, after the implementation of several management practices recommended by the TMDL at a monitoring location near the watershed outlet, the number of samples exceeding the newly-established WQC of 35 mg/L decreased from 71 % to 40 % (Utah DEQ, 1998). There remains, however, uncertainty in this single value of percent exceedance (i.e., 40 %) since no statement of measurement error was given. Additionally little information was provided about the number of samples associated with the new exceedance percentage or the time period

over which they were collected, leading to uncertainty about the representativeness of the samples from which the 40 % exceedance was derived.

The purpose of this study was to quantify uncertainty associated with in-stream continuous turbidity measurements, analytical methods of total suspended solids (TSS) concentration determination and uncertainty associated with the regression between turbidity and TSS. The combined uncertainty was then applied to regression-computed TSS concentrations to calculate prediction intervals that provide context for using the continuous TSS estimates in determining compliance with the WQC set forth by the TMDL study that established a target TSS concentration of 35 mg/L in sections of the Little Bear River.

5.2 Study Area

The Little Bear River watershed is located in the south end of Cache Valley, Utah and encompasses roughly 740 km^2. The headwaters form in the Bear River Mountain Range, and the river drains to Cutler Reservoir on the west side of Cache Valley. Elevations within the watershed range from 1,345 m to 2,865 m. Land use within the upper watershed is comprised mostly of forest and range (grazing) lands, while the lower portion of the watershed is made up primarily of agricultural lands and some urban development. Hydrology of the watershed is dominated by spring snowmelt, with water in the upper watershed being stored within two reservoirs, Porcupine Reservoir in the upper East Fork and Hyrum Reservoir on the mainstem, before being released and diverted into irrigation canals for agricultural use during the irrigation season.

Since 2005, research conducted by the Utah Water Research Laboratory at Utah State University has facilitated the installation of two continuous water quality monitoring stations (Figure 5.1). Table 5.1 summarizes the locations and descriptions of the water quality monitoring sites located in the watershed along with the dates from which the respective measured parameters are available. One of these stations (site 1) is located in the upper watershed above Hyrum reservoir, where stream channels are relatively steeper and sometimes have velocities as high as 2.0 m/s. The lower portion of the watershed includes the second monitoring site at the terminus of the watershed near the confluence with Cutler Reservoir. In this lower section of the river, channel slopes are lower and velocities are almost always < 0.7 m/s. The river also exhibits a more meandering planform, and soils are predominantly fine grained lacustrine sediments.

A numeric WQC for the Little Bear River was established in a TMDL study that identified two reaches of the river where concentrations of total suspended solids (TSS) were to be lowered to be no greater than 35 mg/L. The first reach is in the lower watershed, covering the 45 km reach from monitoring site 1 upstream to Hyrum Reservoir. The second reach, covering 11 km and containing site 2, is from Hyrum Reservoir upstream to the confluence with the East and South Forks near the town of Avon as shown in Figure 5.1 (Utah DEQ, 1998).

5.3 Methods

Turbidity, water temperature and gage height were measured using permanently installed, *in situ* sensors every 30 minutes at both sites in the Little Bear River. Turbidity and water temperature were measured using a Forest Technology Systems DTS-12

turbidity sensor. The DTS-12 uses a laser diode with a wavelength in the near-infrared (780 nm) spectrum and measures light scatter at 90 degrees to the incident beam (Forest Technologies Ltd, 2007). This geometry results in measurement units of Nephelometric Turbidity Units (NTU), and the probe has a range from 0-1600 NTU. Accuracy of the probe is $\pm$ 2 % for turbidity values 0-399 NTU and $\pm$ 4 % for values of 400-1600 (Forest Technology Systems Ltd., 2007). Each turbidity measurement consists of 100 instantaneous observations made over a five second time interval, after which summary statistics are output and recorded every 30 minutes. From the 100 observations, the median was used as the "observed" turbidity value throughout this study. Temperature was measured with the DTS-12 from an internal encapsulated thermistor and used to compensate turbidity measurements. The uncertainty in turbidity measurements was identified from manufacturer accuracy specifications. However, because the uncertainty associated with the accuracy of the turbidity measurement that corresponds to the WQC (35 mg/L) was judged to be negligible ($\pm$0.5 NTU), it was not accounted for in the calculation of the 90 % prediction intervals for regression computed TSS concentrations or in estimates of percent exceedance of the WQC value.

A total of 136 and 241 surface grab samples were collected from August 2005 to May 2012 and October 2007 to May 2012 for sites 1 and 2 respectively. Total suspended solids concentrations in stream grab samples were determined using Standard Methods 2540D, Total Suspended Solids Dried at 103-105° C (APHA, 1995) and EPA method 340.2, Total Suspended Solids, Mass Balance or EPA method 160.2 Residue Nonfilterable Total Suspended Solids. Laboratory duplicates were performed on at least

10 percent of the samples collected in the field. Analytical uncertainty in TSS concentration was estimated based on the percent differences between duplicate samples that were analyzed in the lab. Percent differences were calculated by dividing the difference between TSS concentrations for a pair laboratory duplicates by the average of the two samples and then multiplying by 100. The overall analytical error for each site was calculated by averaging all of the percent difference values determined from the laboratory duplicate measurements of TSS.

Procedures for development of regression relationships between turbidity and TSS are described in Section 3.3.2 of this thesis. Uncertainty associated with regression computed TSS concentrations was determined using prediction intervals. These prediction intervals were calculated for each value in the estimated TSS time series using the following equation (Helsel and Hirsch, 2002):

$$E(y_i) \pm t_{(\alpha/2, n-1)} \times s \sqrt{1 + \frac{1}{n} + \frac{(x_i - \bar{x})}{\sum_{i=1}^{n}(x_i - \bar{x})^2}} \qquad \text{Eqn. 5.1}$$

where $E(y_i)$ is the estimated TSS concentration, t is the t-statistic from the student's t distribution where $\alpha = 0.10$ for 90 % confidence, s is the standard error of the regression or RMSE, n is the number of paired turbidity-TSS observations, x_i is the observed turbidity, $\bar{x}$ the is the mean of the observed turbidity values and i is an index for the number of paired observations where $i = 1,2,\ldots n$.

Each of the prediction interval values calculated using Equation 5.1 was then converted to a percent by dividing them by the mean observed TSS concentration. This was done to scale the error over the large (observed) TSS range (1-2450 mg/L). The

analytical uncertainty of TSS determination was added to the 90 % prediction interval,

and the sum was used as the combined uncertainty for regression computed TSS values.

5.4 Results and Discussion

Results presented in this section are in the context of the WQC established by the

Little Bear River TMDL (Utah DEQ, 1998). The continuous time series of regression

computed TSS concentrations were compared to the WQC of 35 mg/L to show how the

techniques presented in this study can be applied. Summary statistics for the regression

computed TSS concentrations from the two continuous water quality monitoring sites in

the Little Bear River watershed are shown in Tables 5.2 and 5.3. The summaries include

yearly totals for the number of data points (*n*), the mean, median, standard deviation,

maximum and minimum for half-hourly regression-computed TSS concentrations for six

years, with each site having greater than 90000 regression-computed TSS values.

The average uncertainty in analytical TSS determination was 9.9 % and 8.2 % for

sites 1 and 2 respectively. The uncertainty associated with the least squares regression

development of turbidity and TSS was determined by the 90 % prediction intervals that

are 43 % (± 48.2 mg/L) for site 1 and 44 % (±13.8 mg/L) for site 2 for each of their

respective predicted TSS concentrations. The combined uncertainty—from analytical

and regression procedures—was ±52.7 % and ±52.2 % of a given estimated TSS

concentration for sites 1 and 2, respectively.

Tables 5.4 and 5.5 list the percent exceedance values for both the regression

computed TSS concentrations and the upper and lower limits of the 90 % prediction

intervals (percent exceedance ranges) for the years 2006-2011 for both sites. For all

years combined, 20 % of all regression-computed TSS concentrations at site 1 were

greater than the WQC. However, after incorporating the combined uncertainty, it can be

stated with 90 % confidence that 11-29 % of all regression computed TSS values

exceeded the WQC at site 1. For site 2, 44 % of all regression computed TSS

concentrations were greater than the WQC. After incorporating the combined

uncertainty, it can be stated with 90 % confidence that 1-68 % of all regression computed

TSS values exceeded the WQC at site 2.

The estimated percent exceedance ranges are very different across the two sites.

Percent exceedance values at site 2 were greater than those at site 1; however, site 2

demonstrated significantly more uncertainty in the estimation the exceedance of the

WQC, as demonstrated by the much wider range of percent exceedance values

represented by the upper and lower limits of the 90 % prediction intervals. This is

demonstrated graphically in Figures 5.2 and 5.3, which show TSS concentration duration

curves, along with their respective prediction intervals, for the year 2009 for both sites.

These figures show that estimated TSS values reach much higher levels at site 1 (e.g., >

1000 mg/L) but for short durations, whereas estimated TSS concentrations at site 2

remain in the range surrounding the 35 mg/L criterion value (e.g., between 20 and 50

mg/L) for much of the year.

Interannual variability in percent exceedance values was also significant at both

sites. Tables 5.4 and 5.5 show percent exceedance values for regression computed TSS

concentrations as high as 41 % in 2011 and as low as 9 % in 2007 at site 1 and as high as

53 % in 2010 and as low as 32 % in 2011 at site 2. Figures 5.4 and 5.5 show TSS

concentration duration curves with corresponding prediction intervals for the year 2011

for sites 1 and 2, respectively. For site 1 it is estimated that 28-49 % of all TSS

concentrations were greater than WQC in 2011. The percent exceedance interval shifted

significantly to higher percent exceedance values compared to year 2009 (Figure 5.2).

This is because the 2010-2011 winter season brought deeper snow pack in the mountains

resulting in higher stream flows and sediment fluxes during the spring snow melt. The

maximum discharge rates at site 1 were 19 and 64 m^3/s for the years 2009 and 2011

respectively. At site 2, it is estimated that 0.5-63 % of all TSS concentrations were

greater than the WQC. The maximum discharge rate at site 2 was slightly higher in 2011

than 2009 (11 compared to 13 m^3/s), but the resulting TSS concentrations were lower

than 2009 as shown in Tables 5.4 and 5.5.

Table 5.6 compares percent exceedance values of traditional monitoring

techniques (grab samples) and regression computed TSS concentrations. For the year

2008 at site 1, TSS concentrations from 80 grab samples collected throughout the year

were determined. The percent of grab samples exceeding the WQC was 36 % while the

percent exceedance of the regression computed TSS concentrations was only 16 % (9-24

% based on the prediction intervals). For site 2, a similar difference was observed in the

year 2006, where 58 % of the grab samples (52 samples analyzed) exceeded the WQC,

but only 36 % of the regression computed TSS concentrations (0.3-67% based on the

prediction intervals) exceeded the criteria. At site 2, the percent exceedance of the grab

samples falls within the range of percent exceedances given by the 90 % prediction

intervals, but at site 1 the percent exceedance calculated from the grab samples lies

significantly outside the range of percent exceedances given by the 90 % prediction intervals.

5.5 Conclusions

High frequency estimates of TSS concentrations derived from turbidity can aid in determining compliance with a WQC or whether water quality goals are being met at a particular site. In this paper, this was demonstrated by quantifying the uncertainty associated with continuous, regression computed TSS concentration estimates, and then determining how that uncertainty translates into a range of potential percent exceedances of the WQC. Results varied at the two sites, with much wider percent exceedance ranges at the lower watershed site versus the upper watershed site. When compared with percent exceedance values computed from grab samples, results from the continuous TSS estimates made over the same time period of the grab samples were significantly different, although the percent exceedance of grab samples at site 2 did fall within the relatively wide range of potential percent exceedance values given by the limits of the 90 % prediction intervals.

A significant advantage of using continuous TSS estimates to assess compliance is that, because continuous TSS concentration estimates can cover a wide time range with high temporal resolution, they are better able to capture the full range and timing of TSS concentrations than sporadic grab samples. However, because there is error in both measurement of the regression variables (i.e., turbidity and TSS) and in the regression itself, it is necessary to quantify the uncertainty in the estimated TSS values and translate it into a subsequent uncertainty in the estimate of the percent of time that a given criterion

value is exceeded. The results above show how the percent exceedance can be expressed as a range of potential exceedance percentages that account for the combined uncertainty in the continuous TSS estimates. By doing so, a range of potential percent exceedance values can be presented that reflects the full range of conditions at a site and accounts for uncertainty introduced through the use of surrogate methods.

References

American Public Health Association (APHA). (1995). "Standard methods for the examination of water and wastewater." American Public Health Association, American Water Works Association, and Water Pollution Control Federation, Washington, DC.

Bertrand-Krajewski, J. L. (2006). "Uncertainties in measured data in urban drainage." *Proceedings of Integrated Urban Water Management Modelling: Challenges and Developments, eWater CRC*, Canberra, ACT, Australia, 17-20.

Forest Technology Systems, Ltd. (2007). DTS-12 SDI-12 Turbidity sensor operating manual. Forest Technology Systems, Victoria, BC.

Gippel, C. J. (1995). "Potential of turbidity monitoring for measuring the transport of suspended solids in streams." *Hydrological Processes*, 9(1), 83-97.

Helsel, D. R., and Hirsch, R. M. (2002). "Statistical methods in water resources." *Technometrics*, 323.

Jastram, J. D., Zipper, C. E., Zelazny, L. W., and Hyer, K. E. (2010). "Increasing precision of turbidity-based suspended sediment concentration and load Estimates." *Journal of Environmental Quality*, 1306.

Jones, A. S., Stevens, D. K., Horsburgh, J. S., and Mesner, N. O. (2010). "Surrogate Measures for providing high frequency estimates of total suspended solids and total phosphorus concentrations." *Journal of the American Water Resources Association*, 47(2), 239-253.

Minella, J. P. G., Merten, G. H., Reichert, M., and Clarke, R. T. (2008). "Estimating suspended sediment concentrations from turbidity measurements and the calibration problem." *Hydrological Processes*, 1830(August 2007), 1819-1830.

Rasmussen, P. P., Gray, J. R., Glysson, G. D., and Ziegler, A. C. (2009). "Guidelines and procedures for computing time-series suspended-sediment concentrations and loads from in-stream turbidity-sensor and streamflow data." U.S. Geological Survey Techniques and Methods book 3, chap. C4, 53 p.

Utah Department of Environmental Quality (Utah DEQ). (1998). "Little Bear River Watershed TMDL." < http://www.waterquality.utah.gov/TMDL/Little_Bear_River _TMDL.pdf> (April 2012).

Table 5.1. Summary of water quality monitoring sites in the LBR watershed. The date range reflects the time period from which the continuous monitoring began

Site	Site Name	Date Range	Latitude	Longitude	Site Description
1	Paradise	June 2005-Present	41.5756	-111.8552	Located a few miles upstream of Hyrum reservoir
2	Mendon	August 2005-Present	41.7185	-111.9464	Located near the boundary of the watershed at Cutler reservoir

Table 5.2. Summary statistics for regression computed TSS concentrations (mg/L) for years 2006 - 2011 and all years aggregated at site 1. n is the number of half-hourly computed TSS concentrations and SD is the standard deviation

Year (Jan-Dec)	n	Mean	Median	SD	Max.	Min.
2006	17519	45.7	13	146.3	2962	2.6
2007	17411	22.9	10.6	67.4	2150	3
2008	17143	31.9	12.7	85.2	2419	2.5
2009	17257	27.5	10.6	69.5	1770	1.7
2010	16505	28.7	11.4	79.1	2064	1.7
2011	16457	79.7	21.9	172.7	2784	2.1
All Years	102283	39.1	12.5	112.1	2962	1.7

Table 5.3. Summary statistics for regression computed TSS concentrations (mg/L) for years 2006 - 2011 and all years aggregated at site 2. n is the number of half-hourly computed TSS concentrations and SD is the standard deviation

Year (Jan-Dec)	n	Mean	Median	SD	Max.	Min.
2006	17519	29.2	30.5	13.4	96.6	3.8
2007	17147	34.7	34.4	18.6	89.5	3.6
2008	17240	33.5	31.7	18.5	91.1	4.4
2009	17289	31.8	32.9	16.5	86.7	5.6
2010	16402	35.9	36.1	17.7	101.6	4.4
2011	14294	28.9	26.5	14.3	84.5	5.2
All Years	99883	32.4	31.9	16.9	101.6	3.6

Table 5.4. Percent exceedance values and 90% prediction interval (percent exceedance) ranges for TSS concentration at site 1

Year (Jan-Dec)	% Exceeding 35 mg/L		
	% Exceedance	Upper P.I.	Lower P.I.
2006	23.0	32.6	11.6
2007	8.9	18.7	3.9
2008	16.3	24.2	8.5
2009	15.1	24.2	6.7
2010	16.6	25.9	6.4
2011	41.3	48.8	27.9
All Years	20.0	28.9	10.7

Table 5.5. Percent exceedance values and 90% prediction interval (percent exceedance) ranges for TSS concentration at site 2

Year (Jan-Dec)	% Exceeding 35 mg/L		
	% Exceedance	Upper P.I.	Lower P.I.
2006	35.5	68.9	0.3
2007	49.0	70.5	2.0
2008	44.2	66	1.2
2009	46.1	66.2	0.1
2010	53.0	73.3	1.7
2011	32.1	62.9	0.5
All Years	43.5	68.1	1.0

Table 5.6. Comparison of percent exceedance between grab samples and continuous estimates of TSS concentration for sites 1 and 2

Site	Type	Year	% Exceed	Count
1	Grab Sample	2008	36.3	80
1	Continuous	2008	16.3	>17,000
2	Grab Sample	2006	57.7	52
2	Continuous	2006	35.5	>17,000

Figure 5.1. Location of Little Bear River watershed water quality monitoring sites
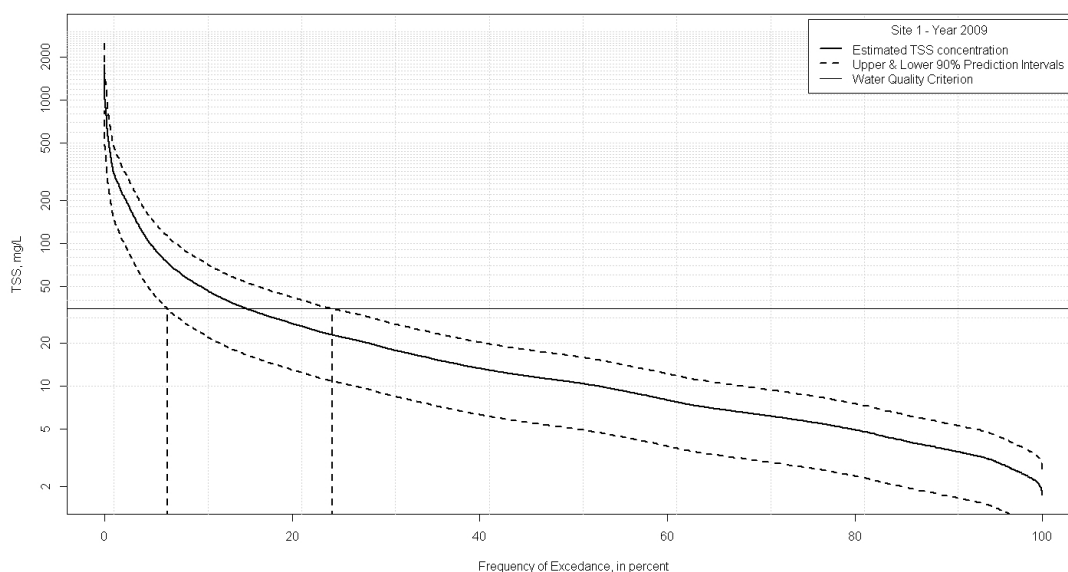
Figure 5.2. TSS concentration duration curve showing the upper and lower 90% prediction intervals for the year 2009 at site 1. The solid, dark horizontal line represents the WQC of 35mg/L. The dashed vertical lines intersect the x-axis at points that represent the percent exceedance of the WQC with 90% confidence
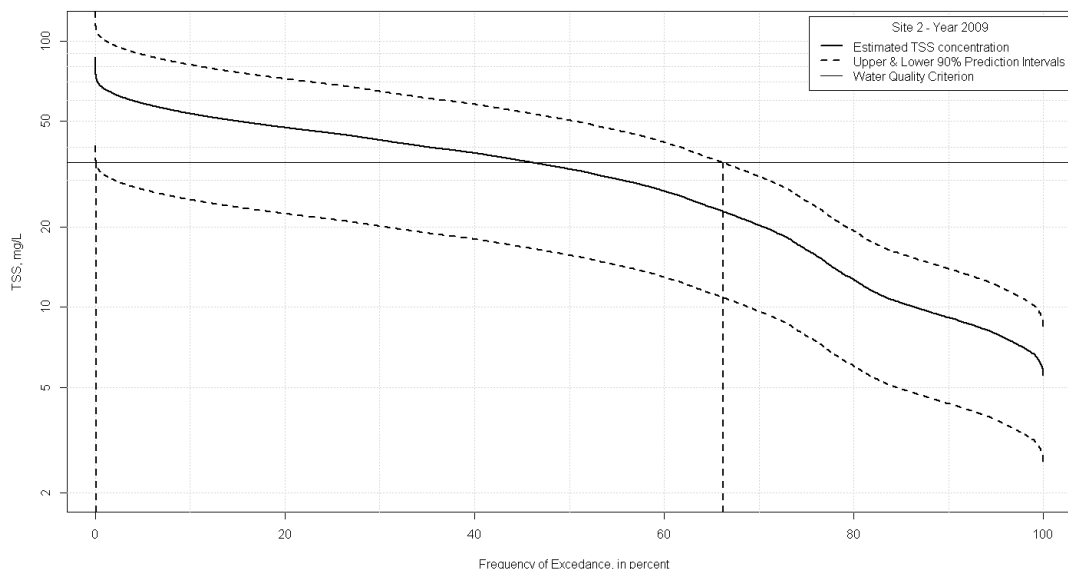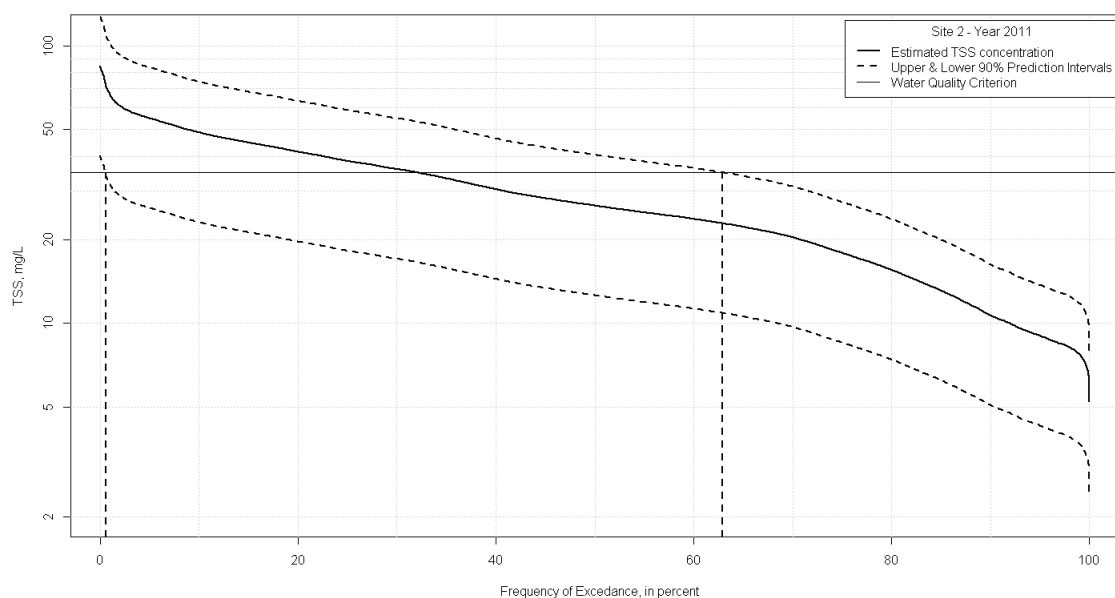

Figure 5.3. TSS concentration duration curve showing the upper and lower 90% prediction intervals for the year 2009 at site 2. The solid, dark horizontal line represents the WQC of 35mg/L. The dashed vertical lines intersect the x-axis at points that represent the percent exceedance of the WQC with 90% confidence
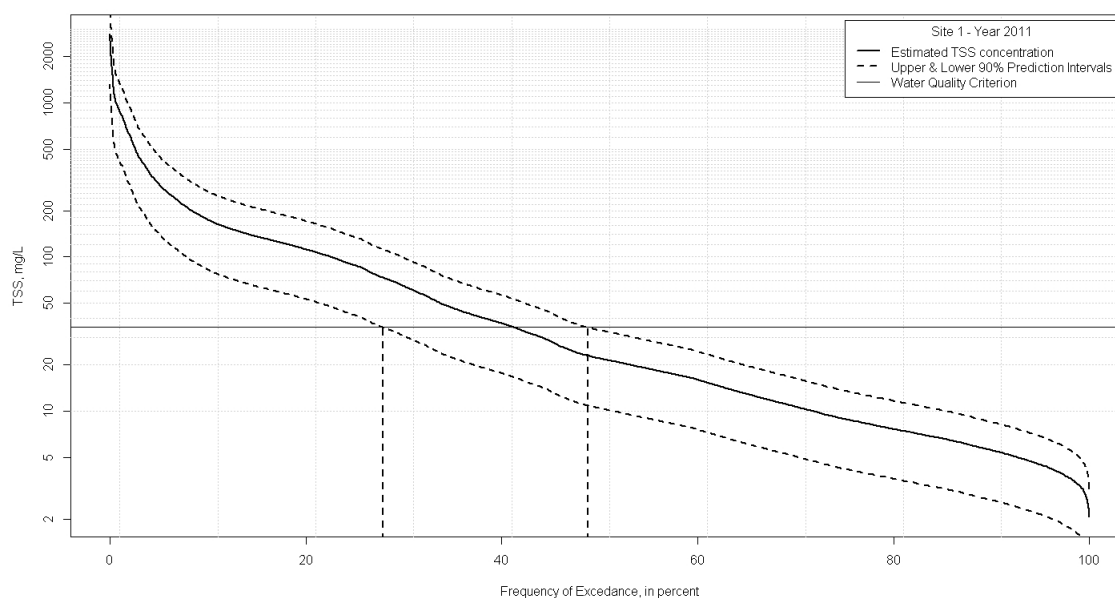
Figure 5.4. TSS concentration duration curve showing the upper and lower 90% prediction intervals for the year 2011 at site 1. The solid, dark horizontal line represents the WQC of 35mg/L. The dashed vertical lines intersect the x-axis at points that represent the percent exceedance of the WQC with 90% confidence

Figure 5.5. TSS concentration duration curve showing the upper and lower 90% prediction intervals for the year 2011 at site 2. The solid, dark horizontal line represents the WQC of 35mg/L. The dashed vertical lines intersect the x-axis at points that represent the percent exceedance of the WQC with 90% confidence

CHAPTER 6

SUMMARY AND CONCLUSIONS

Regression models to predict TSS concentrations were developed at 6 locations in the Little Bear River watershed. Turbidity and a categorical variable indicating whether turbidity was above or below 40 NTU were found to be significant predictors for TSS concentration at the four locations (sites 1-4) in the upper watershed. Of the two sites (5 and 6) in the lower watershed, turbidity and turbidity squared were significant predictors of TSS concentration at site 6, but at site 5 only turbidity was significant. Four of the six regression equations (sites 2 and 4-6) were very similar when turbidity was less than 40 NTU. While the slope of the regression lines at sites 1-4 continue a steep incline (i.e., slopes >1.1) up to 200 NTU, site 6 in the lower watershed had a decrease in slope.

Particle size analysis demonstrated that turbidity values less than 40 NTU resulted in similar particle size distributions among sites, even though particle concentrations were not similar. Particle size analysis also showed that as turbidities surpassed ~40-60 NTU, particle size distributions at site 6 displayed a distinct shift in the distribution toward smaller particles while the distributions at site 2 showed very little difference from those of the lower turbidity values. Both sites showed similar increases in particle concentration as turbidity increased. This suggests that particle density is the dominate particle property influencing the turbidity-TSS relationship at site 2, while particle size is the dominate property at site 6. Physical attributes related to these two sites, such as steeper channel gradients and higher stream velocities at site 2 and sediment transfer

from upper watershed to lower watershed prevented by Hyrum reservoir, low-slope channel gradients and lower stream velocities at site 6, aide in this evaluation.

These physical attributes, and related particle characteristics, also suggested a connection to the regression model developed at site 6 where the turbidity squared term was found to be a significant predictor with turbidity of TSS concentration. These—along with the regression results—also suggest that the two high turbidity values (139 and 166 NTU) and corresponding TSS concentrations (99 and 137 mg/L) may cause a decrease in slope of turbidity-TSS relationship may have a decreasing slope as suggested by the regression line in Figure 3.2 panel F. The decreasing slope at site 6 was also demonstrated by a comparison with site 2 where the respective TSS concentrations at sites 2 and 6 of 246 and 99 mg/L, (difference of 85%) occurred for similar levels of turbidity (143 and 139 NTU).

Particle size was incorporated into the regression analysis at sites 2, 4 and 6, where it was found to be a significant predictor of TSS on a reduced subset of data for each of these sites. Because fewer, smaller particles pass through site 2, percent concentration (PTC) values for the 1.9-2.25 μm size bin were significant with turbidity in predicting TSS concentration on the full data set.

This research has important application to sediment monitoring programs in rivers and streams where sediment related impairments occur. Surrogate relationships can be used to generate high-frequency estimates of TSS concentrations and loads, which can then be used to assess exceedance of water quality criteria and help better understand the

magnitude and timing of sediment loads. This information can help in targeting the implementation of conservation and restoration efforts.

A second potential way to improved surrogate measures of TSS concentrations is to investigate whether discrete, point-based measures of turbidity and TSS concentration are representative of integrated cross-section averages and to develop a correction factor that accounts for differences. Comparisons between measurements of discrete and cross-section integrated turbidity and TSS concentrations were made at two locations in the Little Bear River watershed. At site 1, differences in turbidity between discrete and integrated turbidity values increased to 22% as turbidity increased, whereas at site 2 differences in turbidity between the two measures remained <5%. When turbidity is used as a surrogate for estimating TSS concentration, differences in turbidity between the two methods may be less important because the parameter of concern is TSS concentration. Correcting for differences between discrete measures and integrated measures of turbidity was not necessary for TSS estimation in this study.

Both monitoring sites demonstrated that for some conditions, a single-point grab sample of TSS can be sufficient to represent the cross-sectional average TSS concentration even though many of the sampling periods (Figure 4.7, sample event 3/23/2012; Figure 4.9, sample events 2/16/2012 and 3/8/2012) showed a high degree variability among verticals. For site 1 the small differences (~7%) between discrete and integrated TSS samples were found for high (246 mg/L) levels of TSS. This level of TSS is significant because it represents the 99th percentile of all observed turbidities. A

correction factor was developed and used to convert discrete TSS concentrations to TSS concentrations that more closely represent integrated cross section averages.

Total annual TSS loads were estimated using the corrected TSS concentrations and the small differences (~7.3%) between annual TSS loads for both water years does not produce strong reason to use a correction factor for lower (<246 mg/L) TSS concentrations. Above this concentration, however, these results show that a correction factor may be most useful for the higher (>246 mg/L) TSS values because a significant portion (21-31%) of the annual TSS load is derived from these higher TSS concentrations. A correction factor for these higher TSS concentrations may be useful to improve quantification of TSS loads in the Little Bear River.

A potential application of the high frequency estimates of TSS concentrations is in determining whether streams are meeting the designated beneficial use or an establish water quality standard. However, uncertainty in the estimates must be quantified to provide the proper context for interpreting the estimates.

Uncertainty in TSS concentrations computed from surrogate measures of turbidity was quantified with statistical confidence. The uncertainty in estimated TSS concentrations was quantified with 90 % confidence from two major sources: analytical TSS determination and least squares regression procedures. The uncertainty in turbidity measurements was identified from manufacture accuracy specifications. However, because the uncertainty associated with the accuracy of the turbidity measurement that corresponds to the WQC (35 mg/L) was judged to be negligible (±0.5 NTU), it was not accounted for in the calculation of the 90 % prediction intervals for regression computed

TSS concentrations or in estimates of percent exceedance of the water quality criterion (WQC).

High frequency estimates of TSS concentrations derived from turbidity can aid in determining compliance with a WQC or whether water quality goals are being met at a particular site. This was demonstrated by quantifying the uncertainty associated with continuous, regression computed TSS concentration estimates, and then determining how that uncertainty translates into a range of potential percent exceedances of the WQC. Results varied at the two sites, with much wider percent exceedance ranges at the lower watershed site versus the upper watershed site. When compared with percent exceedance values computed from grab samples, results from the continuous TSS estimates made over the same time period of the grab samples were significantly different, although the percent exceedance of grab samples at site 2 did fall within the relatively wide range of potential percent exceedance values given by the limits of the 90 % prediction intervals.

A significant advantage of using continuous TSS estimates to assess compliance is that, because continuous TSS concentration estimates can cover a wide time range with high temporal resolution, they are better able to capture the full range and timing of TSS concentrations than sporadic grab samples. However, because there is error in both measurement of the regression variables (i.e., turbidity and TSS) and in the regression itself, it is necessary to quantify the uncertainty in the estimated TSS values and translate it into a subsequent uncertainty in the estimate of the percent of time that a given criterion value is exceeded. The results above show how the percent exceedance can be expressed as a range of potential exceedance percentages that account for the combined uncertainty

in the continuous TSS estimates. By doing so, a range of potential percent exceedance values can be presented that reflects the full range of conditions at a site and accounts for uncertainty introduced through the use of surrogate methods.

CHAPTER 7

ENGINEERING SIGNIFICANCE

It is recognized that suspended sediment plays a critical role in the water quality of rivers and streams. Suspended sediment and its associated pollutants have the potential to negatively impact water for human use and the environment. Improved quantification of suspended sediment concentrations and fluxes at appropriate temporal and spatial scales has important considerations for scientific and engineering related applications which are designed to address current water quality management challenges.

The overall goal of this study was to improve surrogate measures of suspended sediment concentration in order to increase the accuracy and precision of suspended sediment concentrations and fluxes. Improved quantification of surrogate monitoring practices have implications for determining compliance with water quality standards, improved inputs to water quality models and can provide improved understanding of watershed and hydrologic processes for water quality studies.

Increased precision and accuracy of suspended sediment budgets is important to water use, watershed management and policy related decisions because sediment related issues directly impact water quality. Significant resources are dedicated to minimizing pollutant loads/transport and restoring watersheds, and assessing the effectiveness of these efforts is dependent on the accuracy of the field measurements and contextual data analysis. Although fluvial suspended sediment is a natural constituent of steam ecosystems, anthropogenic influences greatly alter natural sediment transport regimes and resulting water quality. Investigation of sediment sources, delivery and transport

processes requires an understanding of dynamic hydrologic processes that vary over space and time scale. The long term, continuous datasets and measurements performed in this research provide a unique opportunity to better inform our understanding of sediment flux in a way that would be impossible with more traditional, low-frequency datasets.

This research also demonstrates the value of quantifying the uncertainty associated with regression computed TSS concentrations and how the uncertainty can be used to provide a more complete approach for assessing compliance with water quality criteria. Quantification of uncertainty in the measurement of water quality parameters is of particular importance given that determination of attainment or exceedance of water quality criteria can have significant economic impacts as policy and regulation of the nation's water resources move and more toward restoring and enhancing water quality. It also serves to reduce the risk associated with compliance decisions that often involves substantial economic investment. This research demonstrates how the quantification of uncertainty in TSS concentrations can be applied to current challenges described in the Little Bear River TMDL study.

This research has important application to sediment monitoring programs in rivers and streams where sediment related impairments occur. This information can help in targeting the implementation of conservation and restoration efforts.

CHAPTER 8

RECOMMENDATIONS FOR FUTURE RESEARCH

The following items were selected as areas for future research in the Little Bear watershed that are aimed at improving estimates of TSS concentration and loads. These areas include ideas to improve the regression analysis performed on existing turbidity and TSS data as well as specific field methods that would validate assumptions of grab samples and to reprove some the conclusion herein to include a more full range of observed data values.

1. Validation of regression models for sites in the little bear river. Many of the sites have over 100 paired turbidity-TSS samples. This number of observations would allow the data set to be split into a training and test portions so that the developed models could be validated. This would be helpful because it could verify the goodness of fit of the model and better quantify the actual error of the predictions.

2. Conduct additional cross section integrated sampling to see if the discrete measures at other sites are representative of the cross section average. This is important because it tests a major assumption of continuous monitoring and grab sample methodologies. There are four other locations in the watershed where this could be carried out. Site 5 (Wellsville is the most logistically feasible location because access to a low-lying bridge is at the location of the existing monitoring station and the stream bed cross section is not obstructed. Perhaps the next most interesting location for conducting discrete vs. cross-section sampling is site 4 (Paradise) since this location experiences the highest discharge (>2000cfs) of all the other sites and it is the last monitoring

station before the contact with Hyrum reservoir. This location presents some difficulty to conduct cross-section integrated sampling, however. This is because the height of the bridge (~25 feet) precludes the use of existing equipment that is not designed or able to collect suspended sediment samples from such heights. Second, and perhaps more limiting, is that on either side of the bridge, the channel cross-section is very much obstructed with large (6 foot diameter) boulders that prevent an accurate measure of water depth across the cross-section. This also inhibits one's ability to distinguish a cross-section that is measureable. A possible option at this site however, is to use an existing cable line that is used by the U.S Geological Survey to make discharge measurements. The cable-reel setup is approximately ~35 feet up stream of the bridge where the U.S.G.S maintains a current stage discharge relationship. Using this cable-reel setup would require a sediment sampler capable of being suspended from a cableway in depths up to 10-12 feet and water velocities up to ~7 ft/s. The DH-59 or DH-95 are potential options to collect the necessary depth-integrated suspended sediment samples. These samplers may cost $700 to $2,200 respectively. More info:

http://water.usgs.gov/osw/pubs/OFR_2005_1087/index.html

3. Conduct further depth-and-width-integrated sampling campaigns at various sites in the Little Bear river watershed when elevated levels of suspended sediment concentration are present. The 2012 spring snowmelt resulted in very low and infrequent elevated turbidity/TSS levels. Although turbidity levels can reach the sensor's maximum at 1600 NTU at site 2 (Lower South Fork), integrated cross-

section averaged measurements have only taken place up to 200 NTU. At site 6 (Mendon) cross-section measurements have been made up to ~20 NTU where the maximum turbidity levels at this site have reached 250 NTU. Depth and width integrated samples should be collected at the higher suspended sediment concentrations in order to verify the appropriateness of the single point grab sample.

4. Compare depth-and-width integrated TSS samples with TSS concentrations estimated from ADCP measurements. During this research, measurements with the ADCP have been made in conjunction with the collection of discrete and width-and-depth-integrated TSS samples at sites 1 and 2 (Lower South Fork and Mendon). It is believe that the necessary data has been collected in order to convert the corresponding acoustic backscatter intensities to TSS concentration for at least two sampling campaigns at each site. Estimating TSS concentration from the ADCP measurements may provide an additional technique that can be used to compare point vs. cross-section averages and to estimate TSS concentrations at a wider range of discharge levels.

5. Carbon analysis on water samples to determine amount of organic matter in TSS samples. Due to laboratory instrument failure, carbon analysis could not be conducted on samples that were analyzed for particle characteristics, i.e., size and concentration. Additional samples could be collected for organic carbon analysis and paired with samples analyzed for TSS and particle size and concentration as these are contributors to turbidity response that is measured in the stream and the specific organic matter content may be a significant factor in predicting TSS concentration.

APPENDIXES

Appendix A. Explanation on the procedure for particle size analysis.

Grab samples for sediment analysis were collected with the purpose of incorporating sediment particle characteristics into turbidity based suspended sediment estimation models in order to reduce variance and improve predictability.

Field Techniques

Depending on sediment loads (determined visually and by turbidity readings) various quantities of stream water were collected from a given site. During high turbidities (>50 NTU), 500 mL to 1000 mL were sufficient for particle characteristic analysis. During times of lower turbidity, 2000 mL or more were necessary. Water samples were collected by dipping an open bottle 8-20 cm below the surface of the water adjacent to existing continuous monitoring turbidity instruments. Sample bottles were kept on ice while in the field and were then refrigerated at 4^0 C for 4-5 days while sediment was allowed to settle.

Instrument Specifications

Particle size and concentration were measured using the LISST-Portable (Laser In-Situ Scattering and Transmissometry) from Sequoia Scientific, which employs laser diffraction techniques and light scattering methods described in AWWA Standard No. 2560D and ISO-13320-1 (Sequoia Scientific, 2010). The range of the LISST-Portable (Type C) is 1.9 - 381μm using the randomly shaped particle inversion method (Agrawal, et. al., 2008). Sample volumes were measured using a graduated cylinder and poured into the 175 mL mixing chamber through a small opening. Once the water level had reached the chamber's capacity, a rubber stopper was inserted into the hole so as to not

allow air bubbles to form inside the chamber. Air bubbles can cause interference with particle sizing and counts as the instrument cannot differentiate between the air bubbles and sediment particles and consequently the bubbles are counted as sediment particles (Sequoia Scientific, 2010). A pump facilitated the continuous mixing of the sample as measurements were taken. Air bubbles can form inside the chamber once the mixer is started. These were removed by turning off the mixer, removing the rubber stopper, reinserting the stopper and starting the mixer again. This process was repeated until air bubbles were no longer visible.

Generally, the LISST-Portable was programmed to measure particle characteristics with a five-second averaging duration. For samples with low sediment concentration, an averaging duration of one minute was used in order to reduce random noise. The LISST-Portable requires the sample to have an optical transmission between 30-90%. The optical transmission is the fraction of laser light that is passing undisturbed through the sample medium (Sequoia Scientific, 2010). Low transmissions prevent the instrument from distinguishing between sediment and random noise. Transmissions greater than 90% inhibit distinction between individual particles. If the optical transmission was lower than 30%, the sample was diluted by removing a given volume (while being mixed) of the solution and adding filtered, deionized water of the same volume until the optical transmission was in the required range. Because of the difficultly in concentrating a sample beyond the initial procedures described above, samples whose optical transmission was at or near 90% were measured with an averaging duration of one minute as recommended by the manufacturer (Sequoia Scientific, 2010).

The LISST-Portable literature explains that when there are particles outside of the measureable range, it will be manifested by a rising tail of particle concentration in the smallest and/or largest size ranges.

Laboratory Procedure

All samples were initially concentrated from the collection bottles (500-2000mL) to 100-500mL by allowing the samples to settle over a period of 4-5 days. In the lab, sample bottles (for a single site) were combined after pouring off the top ~70-80% of the sample volume. Below this level and after the holding time mentioned, it was assumed that all the particles in the measureable range of the LISST were contained. This was tested by retaining the top portion and processing it on the LISST. In these cases, it was found that the optical transmission was too high to be reliable because of the lack of particles present.

Then it was necessary to rinse lingering sediment from the samples bottles using small aliquots of filtered deionized water in order to minimize sediment loss. A volumetric portion of this concentrated sample was then measured and added to the 175 mL LISST sample chamber. Generally, the volume of the concentrated sample added to the chamber was less than 175 mL. This allowed the rinse water from the measuring device to fill the camber to capacity. All aliquots of rinse water were quantified so that the appropriate concentration and dilution adjustments could be made.

Example Data

As an example, a field grab sample of 2020 mL was collected on April 18, 2012 at the Lower South Fork site and was processed in the following way. The 2020 mL

sample was concentrated into a 280 mL sample by allowing it to settle for four days and then pouring it off. The 280 mL sample was then diluted with 110 mL of filtered deionized water in order to rinse lingering particles from the original sample bottle(s). The concentrated sample (280+110) of 390 mL was homogenized and then split into sample A and sample B, 160mL and 230mL, respectively. The 160 mL sample was then diluted to 175 mL in order to run the analysis on the LISST-Portable. The 230 mL sample was concentrated a second time to 150 mL and then diluted to 175 mL. The LISST analysis gives concentrations of 66.2 µl/L for sample A and 88.5 µl/L for sample B. In order to account for these concentrations and dilutions and present the results in context of each other, the following equations were used.

Sample A: Starting concentration = 66.2 μl/L, Volume = 175mL, diluted from 160mL

Applying Equation 1

$$C_1 * V_1 = C_2 * V_2 \qquad \text{Eqn. 1}$$

where C_1=66.2 μl/L, V_1=175 mL and V_2=160 mL. Rearranging and solving for C2

$$66.2\frac{\mu l}{L} * 175\ mL = C_2 * 160\ mL \qquad \text{Eqn. 2}$$

$$C_2 = \frac{66.2\frac{\mu l}{L}*175\ mL}{160\ mL} = 72.4\ \mu l/L \qquad \text{Eqn. 3}$$

gives the resulting concentration of 72.4 μl/L which was corrected for the last dilution.

Applying Equation 1 for the first dilution where volume, V_2, is 114.9 mL.

$$C_2 = \frac{72.4\frac{\mu l}{L}*160\ mL}{114.9\ mL} = 100.9\ \mu l/L \qquad \text{Eqn. 4}$$

V_2 was calculated by first determining the amount of DI water added to the original 280

mL sample that actually went to Sample A (160 mL) and then subtracting this from the

160 mL sample. This was done using Equation 5

$$160\ mL - \frac{160\ mL}{390\ mL} * 110\ mL = 114.9\ mL \qquad \text{Eqn. 5}$$

The final concentration was corrected by dividing the ending concentration (100.9 μl/L)

by the total amount of sample collected.

$$\frac{100.9\frac{\mu l}{L}}{2.02\ L} = 49.95\frac{\mu l}{L} \qquad \text{Eqn. 6}$$

Sample B: Starting concentration = 88.5 µl/L, Volume = 175 mL, diluted from 150 mL

Applying equation 1 from above with data from sample B gives

$$88.5 \frac{\mu l}{L} * 175 \; mL = C_2 * 150 \; mL \qquad\qquad \text{Eqn. 7}$$

where C_1=88.5 µl/L, V_1=175 mL, V_2=150 mL

$$C_2 = \frac{88.5 \frac{\mu l}{L} * 175 \; mL}{150 \; mL} = 103.3 \; \frac{\mu l}{L} \qquad\qquad \text{Eqn. 8}$$

The resulting concentration, correcting for the last dilution gives 103.3 µl/L. Applying

the next dilution using Equation 9

$$C_2 = \frac{103.3 \frac{\mu l}{L} * 150 mL}{230 mL} = \frac{67.3 \mu l}{L} \qquad\qquad \text{Eqn. 9}$$

where volume V_2 is 230 mL. This is the measured amount for sample B.

The next dilution was calculated by first determining the amount of DI water added to the

original 280 mL sample that actually went to Sample B (230 mL) and then subtracting

this from the 230 mL sample. This was done using equation 10

$$230 \; mL - \frac{230 \; mL}{390 \; mL} * 110 \; mL = 165.1 \; mL \qquad\qquad \text{Eqn. 10}$$

The next concentration was calculated using equation 11:

$$C_2 = \frac{67.3 \frac{\mu l}{L} * 230 \; mL}{165.1 \; mL} = 93.8 \; \mu l/L \qquad\qquad \text{Eqn. 11}$$

The final concentration was corrected by dividing the ending concentration (93.8 µl/L) by

the total amount of sample collected.

$$\frac{93.8 \frac{\mu l}{L}}{2.02 \; L} = 46.43 \; \frac{\mu l}{L} \qquad\qquad \text{Eqn. 12}$$

Sample B: Surface Area

Surface area was given in cm^2 as the total surface area for all particles in the 175 mL
sample. To convert this quantity to a per liter basis, Equation 14 was used. For sample A,
the total surface area is 61.6 cm^2 per 175 mL of sample.

$$66.6\frac{cm^2}{175\ mL} * \frac{1000\ mL}{1\ L} = 380.57\frac{cm^2}{L}$$

Eqn. 13

Since a total of 2.02 L were collected in the field, the actual per one Liter basis would be
188.4 cm2/L as given by Equation 14.

$$\frac{380.57\frac{cm2}{L}}{2.02\ L} = 188.4\frac{cm^2}{L}$$

Eqn. 14

Appendix B. Residual plots of regression equations developed at each site.
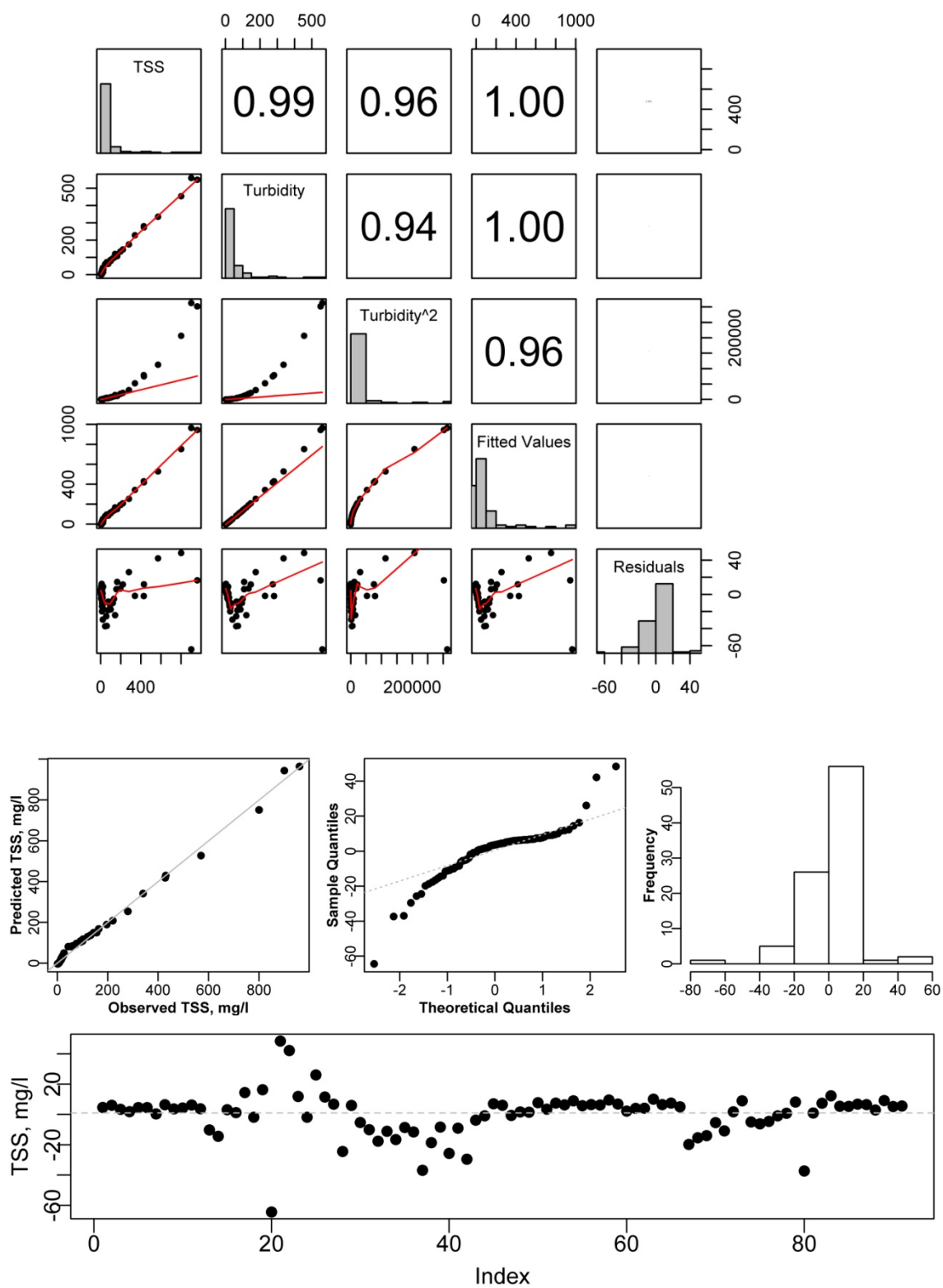
Figure B-1. Residual plots for site 1. The red line is a moving average
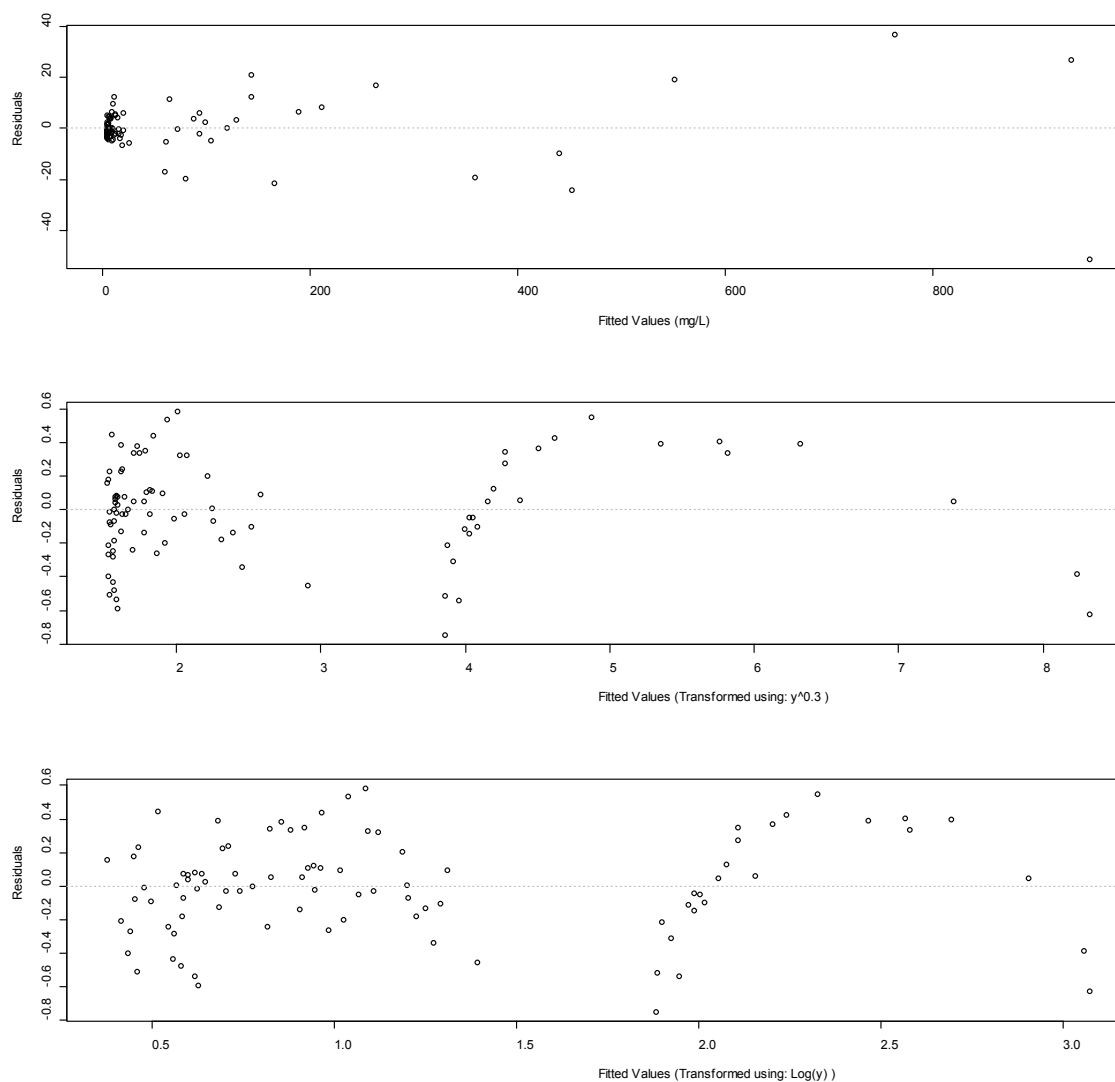
Figure B-2. Comparison of transformations for site 1. The top panel shows the residuals plotted against the fitted (TSS, mg/L) values. The middle panel shows the transformation of $y^{0.3}$ and the bottom panel shows a Log transformation of y
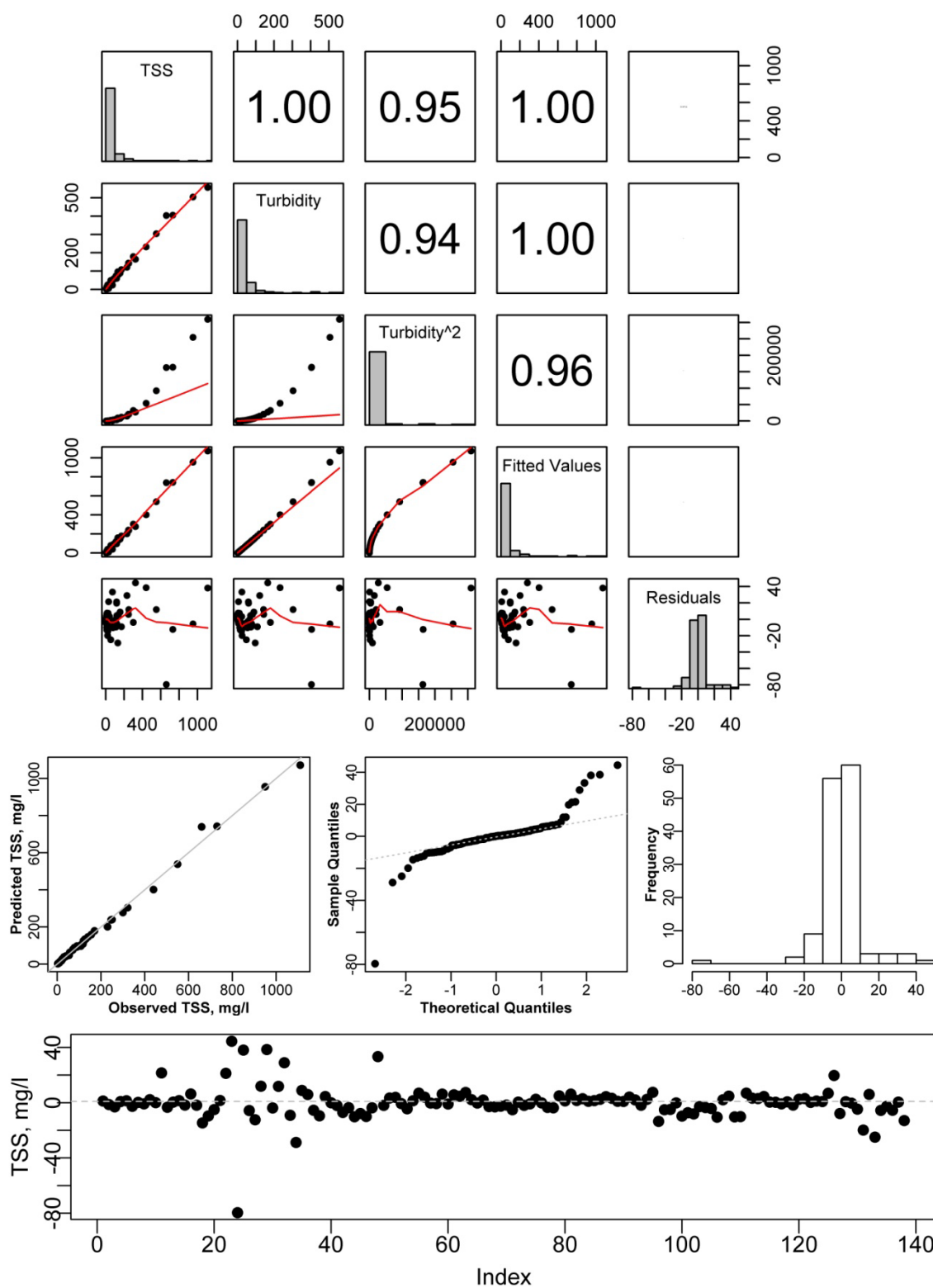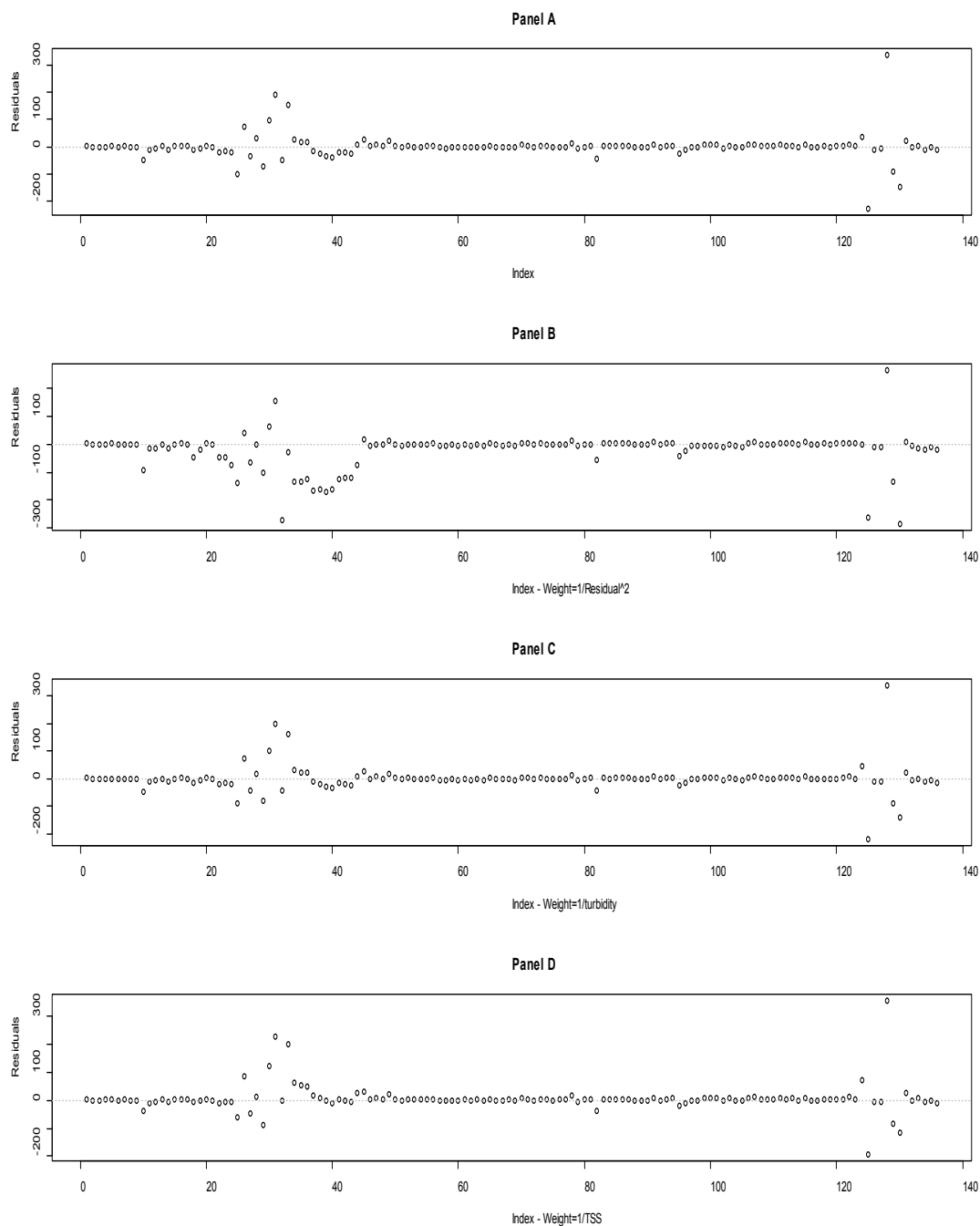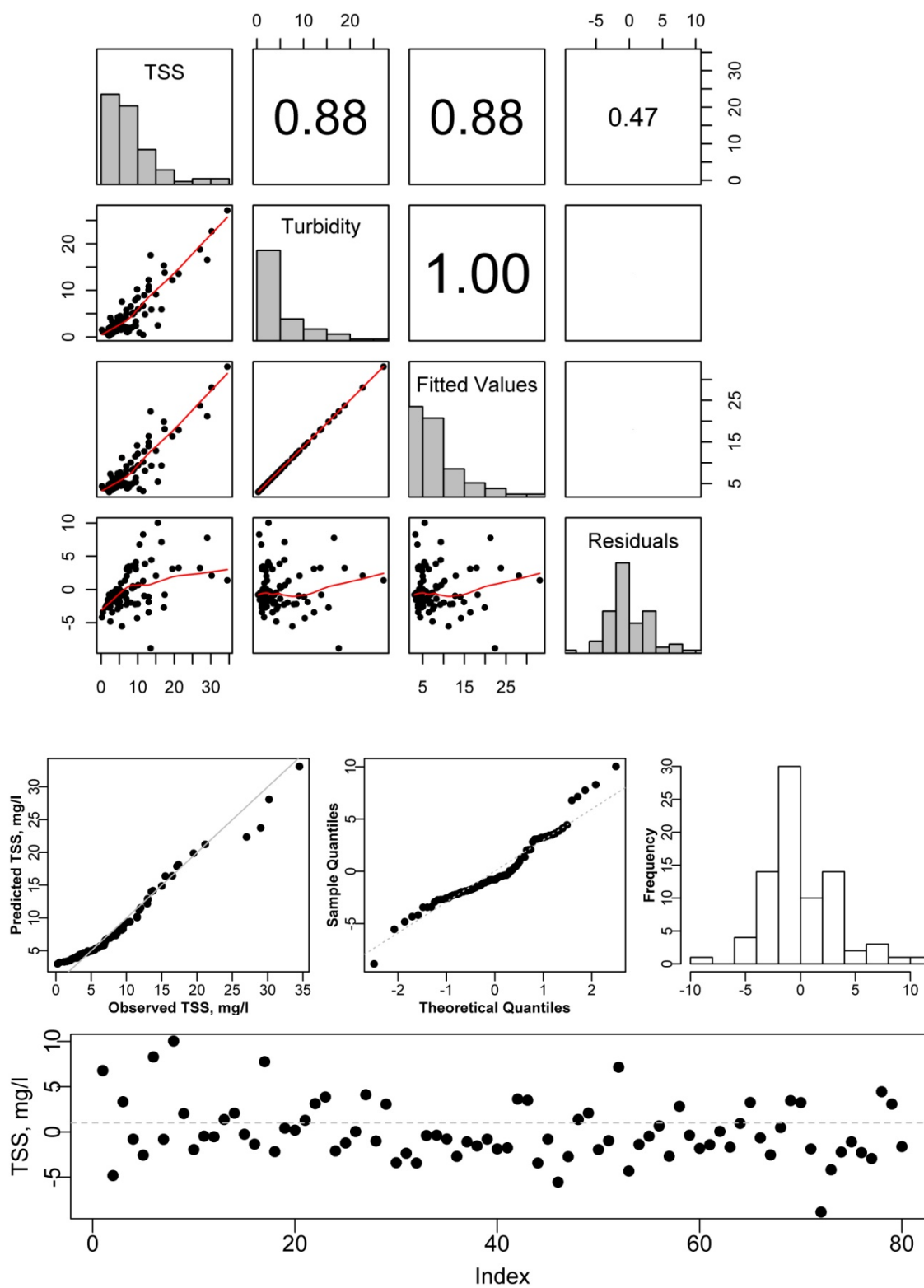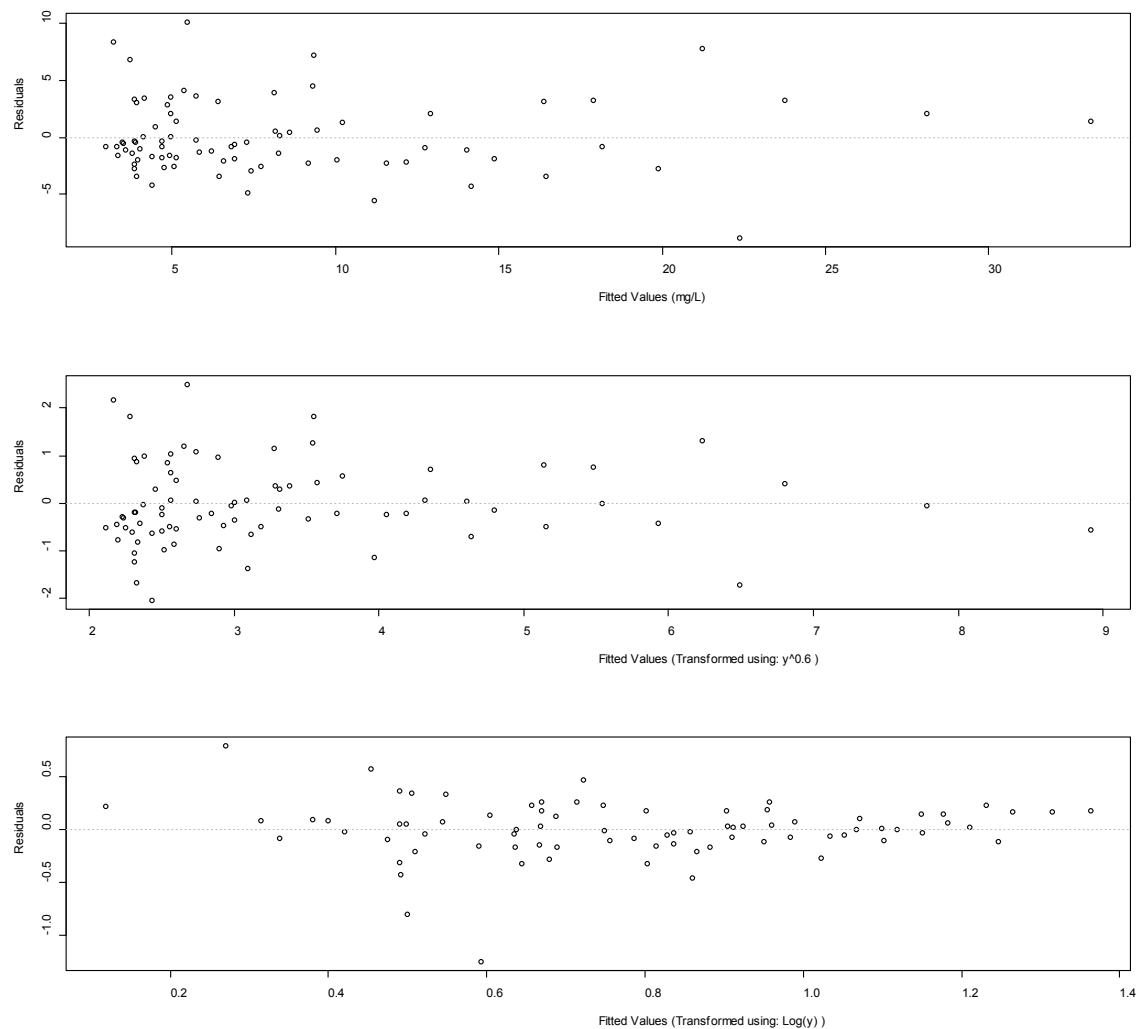
Figure B-3. Residual plots for site 2
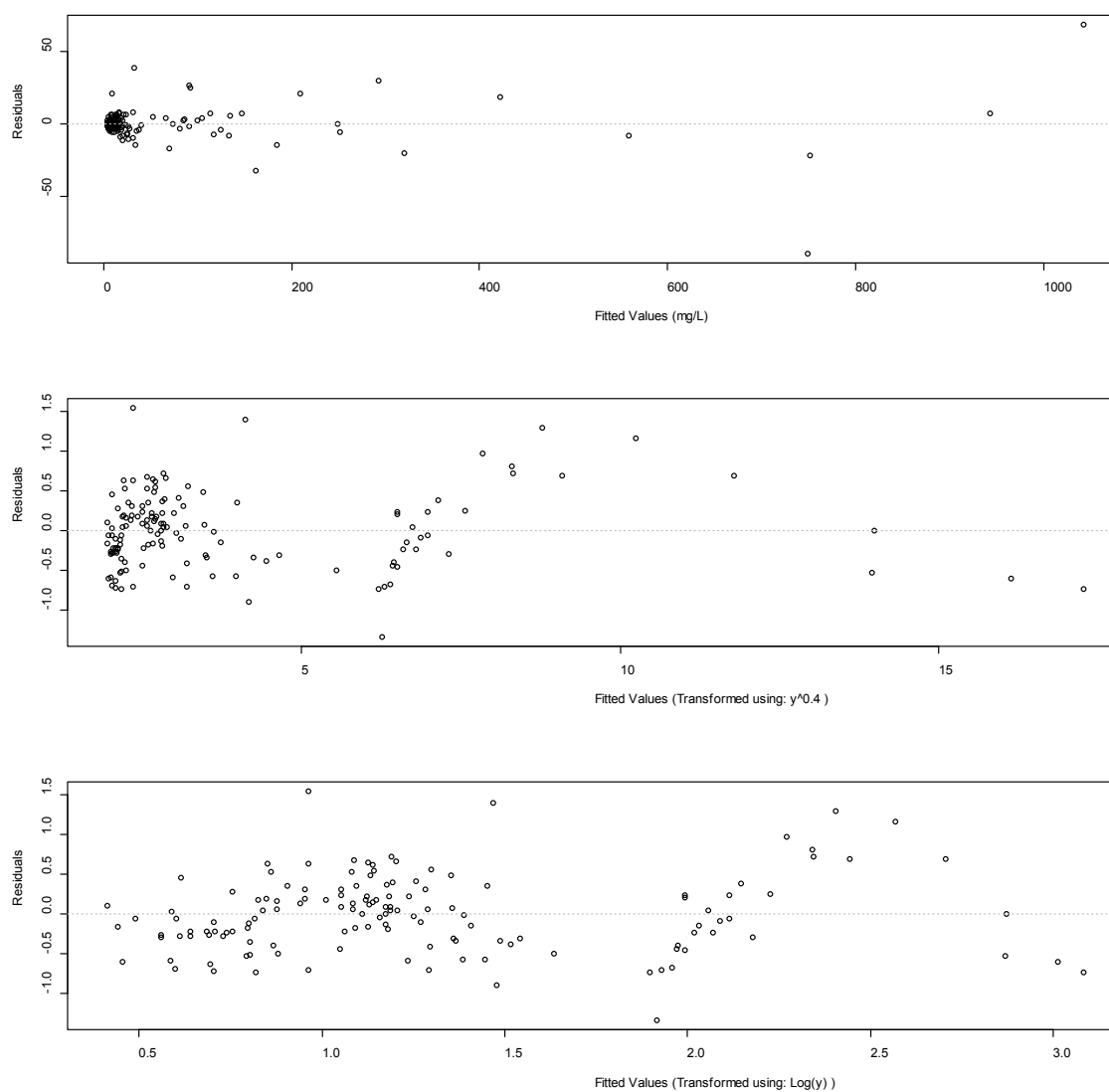
Figure B-4. Comparison of transformations for site 2. The top panel shows the residuals plotted against the fitted (TSS, mg/L) values. The middle panel shows the transformation of $y^{0.4}$ and the bottom panel shows a Log transformation of y
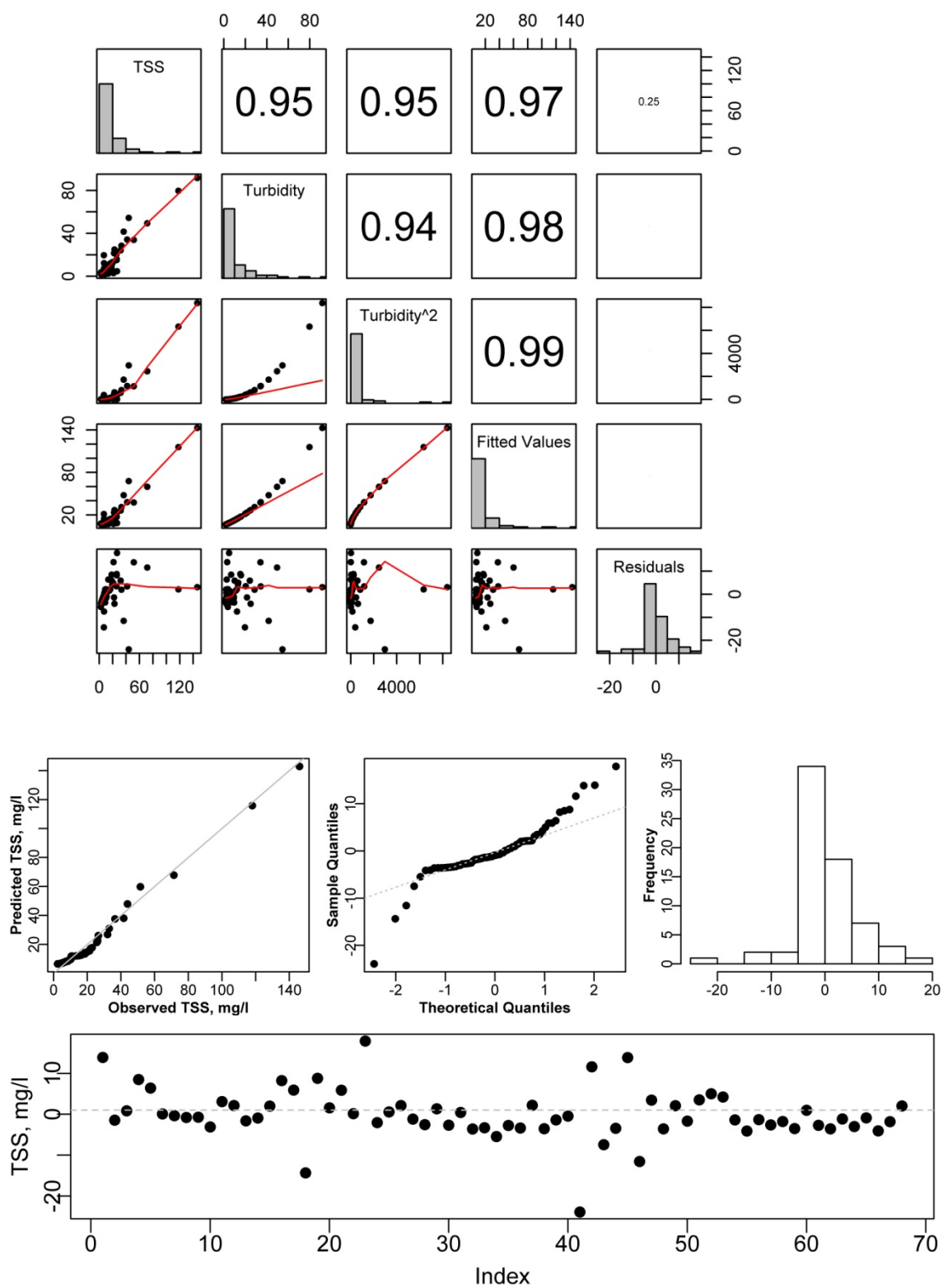
Figure B-5. Residual plots for site 3
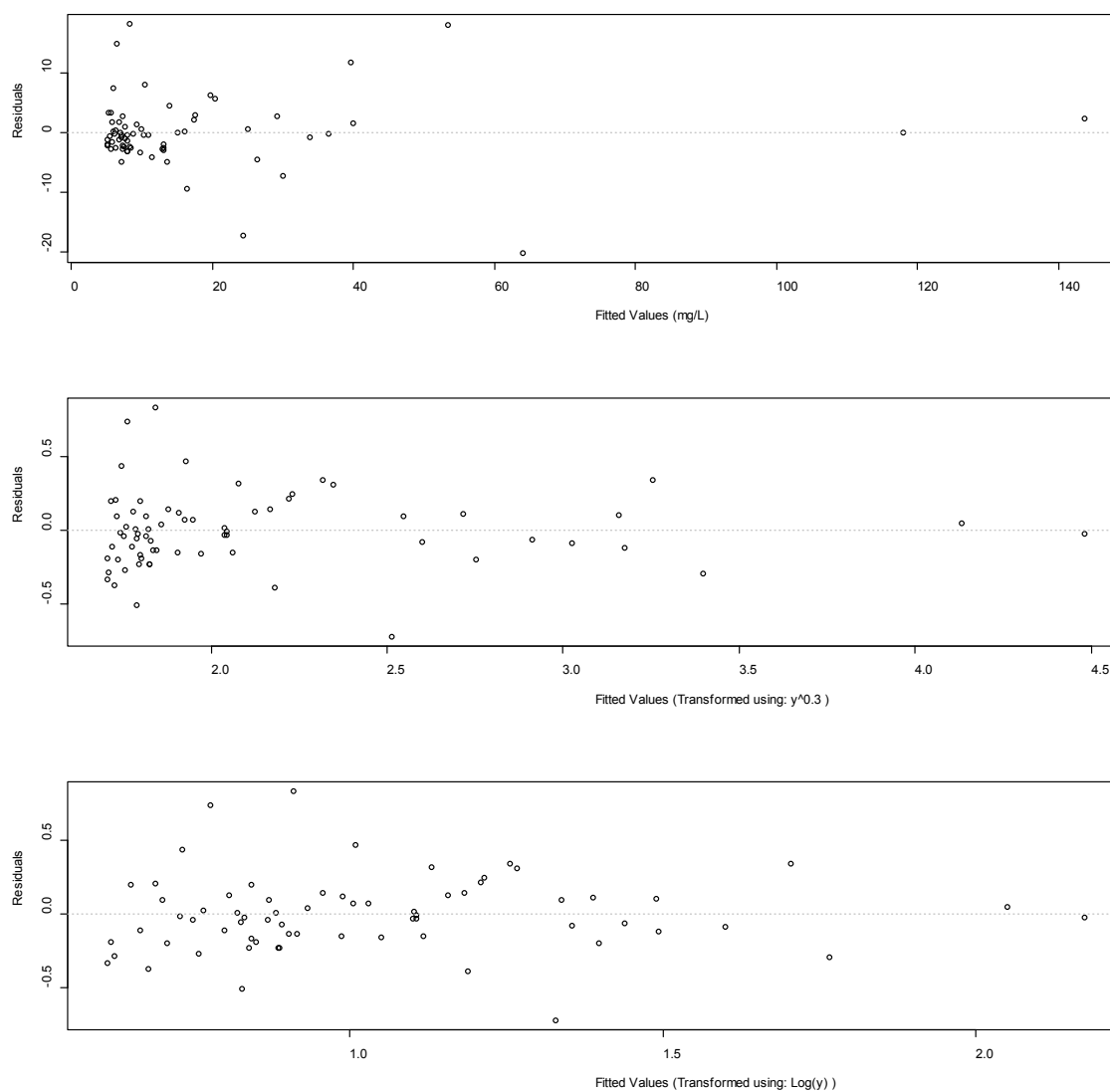
Figure B-6. Comparison of transformations for site 3. The top panel shows the residuals plotted against the fitted (TSS, mg/L) values. The middle panel shows the transformation of y^0.3 and the bottom panel shows a Log transformation of y
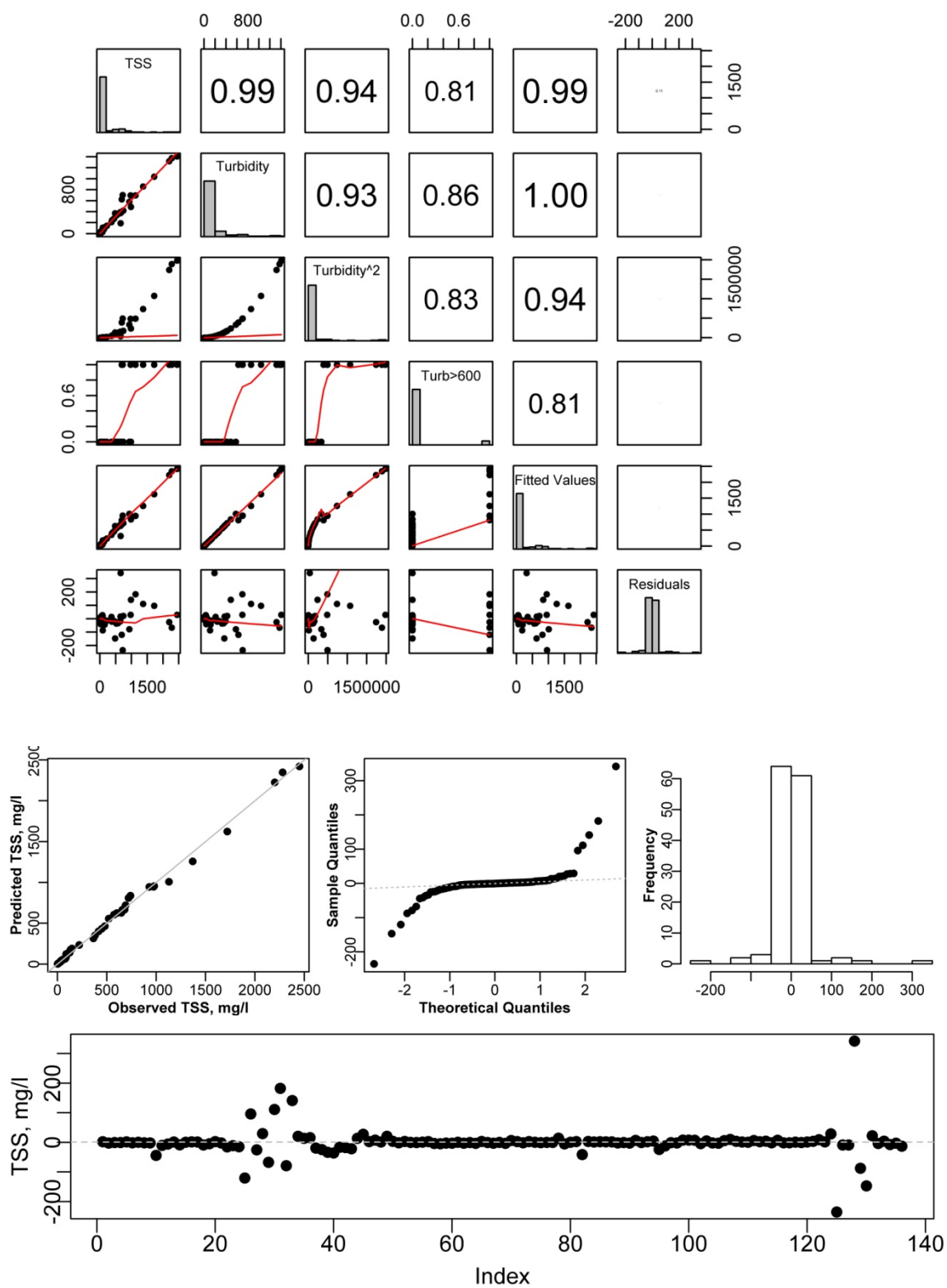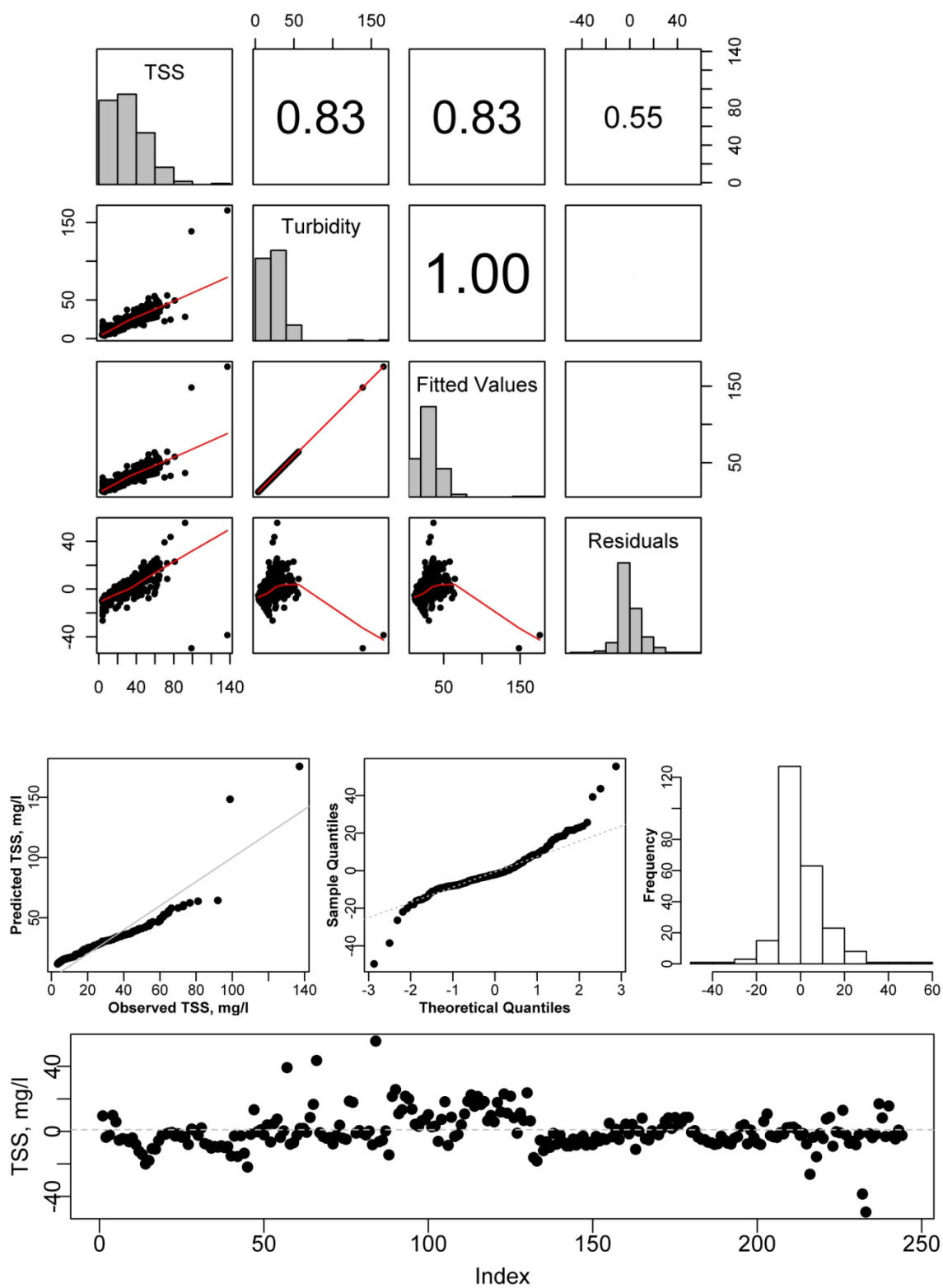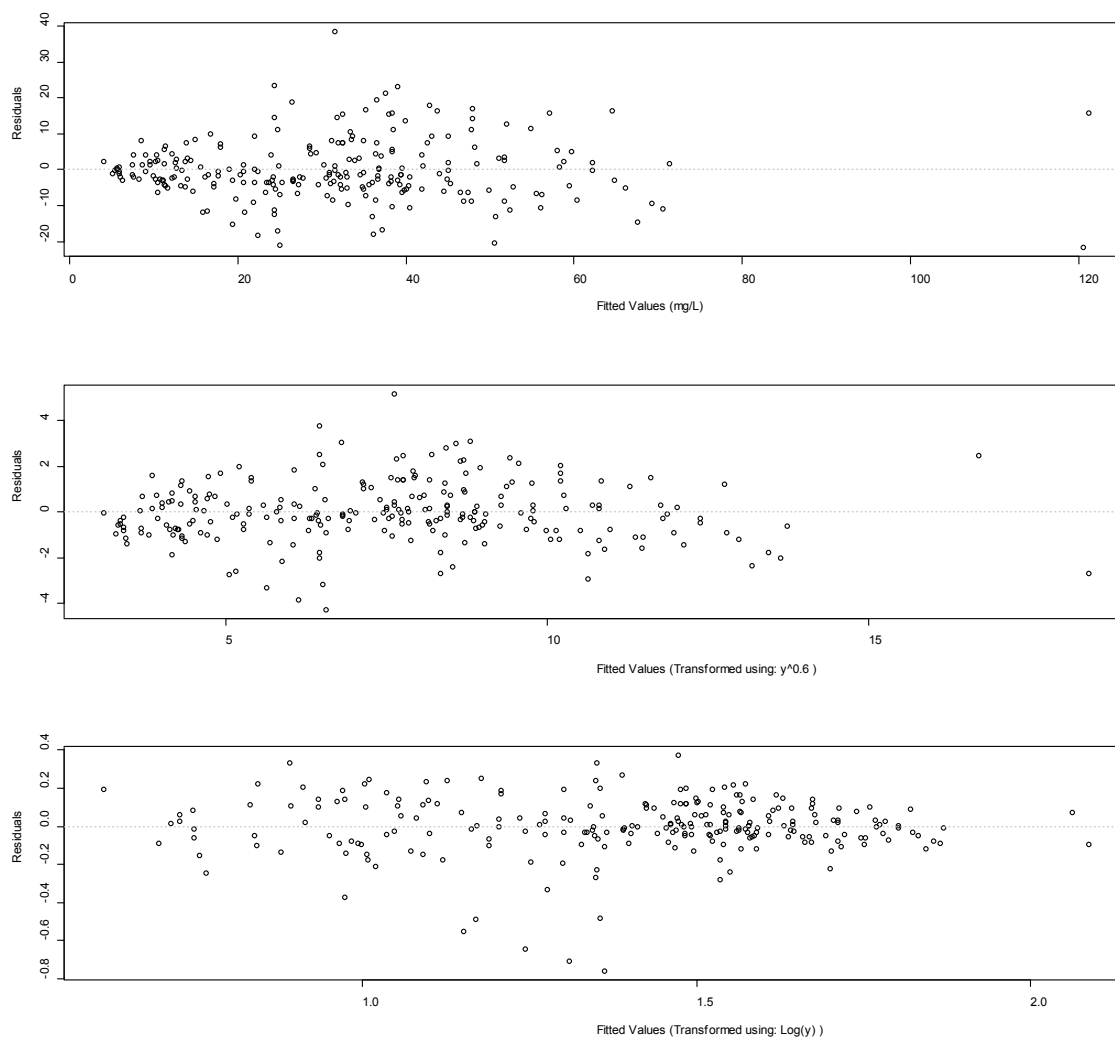
Figure B-7. Residual plots for site 4
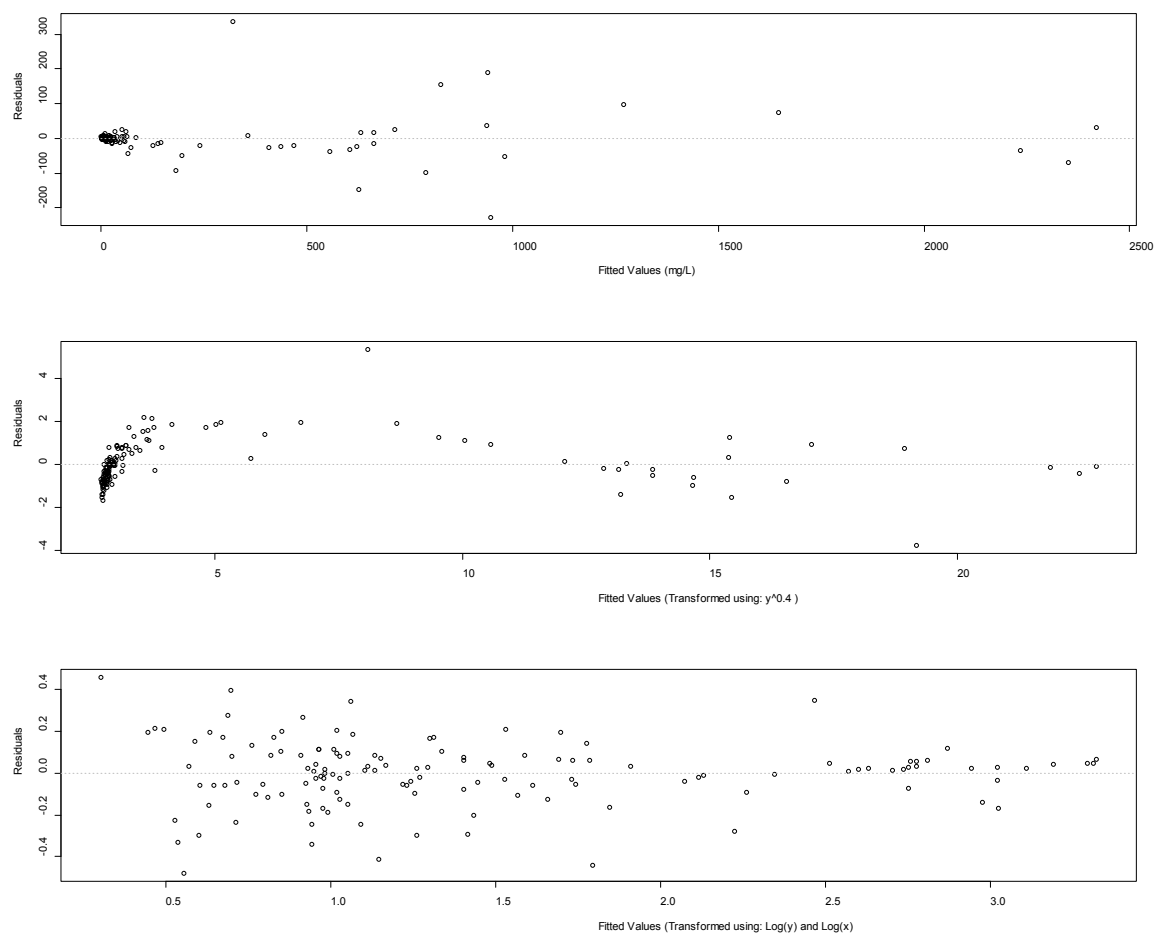
Figure B-8. Comparison of transformations for site 4. The top panel shows the residuals plotted against the fitted (TSS, mg/L) values. The middle panel shows the transformation of y^0.4 and the bottom panel shows a Log transformation

Figure B-9. Comparison of weighted least squares for site 4. Panel A shows the residuals of the regression performed as reported in Table 3.4. Panel B shows the residuals for the regression model using 1/residual^2 as the weight. Panel C shows the regression with 1/turbidity as the weight and Panel D shows the residuals from the regression with 1/TSS as the weight. The index is the order in time

Figure B-10. Residual plots for site 5

Figure B-11. Comparison of transformations for site 5. The top panel shows the residuals plotted against the fitted (TSS, mg/L) values. The middle panel shows the transformation of $y^{0.6}$ and the bottom panel shows a Log transformation of y

Figure B-12. Residual plots for site 6

Figure B-13. Comparison of transformations for site 6. The top panel shows the residuals plotted against the fitted (TSS, mg/L) values. The middle panel shows the transformation of $y^{0.6}$ and the bottom panel shows a Log transformation of y
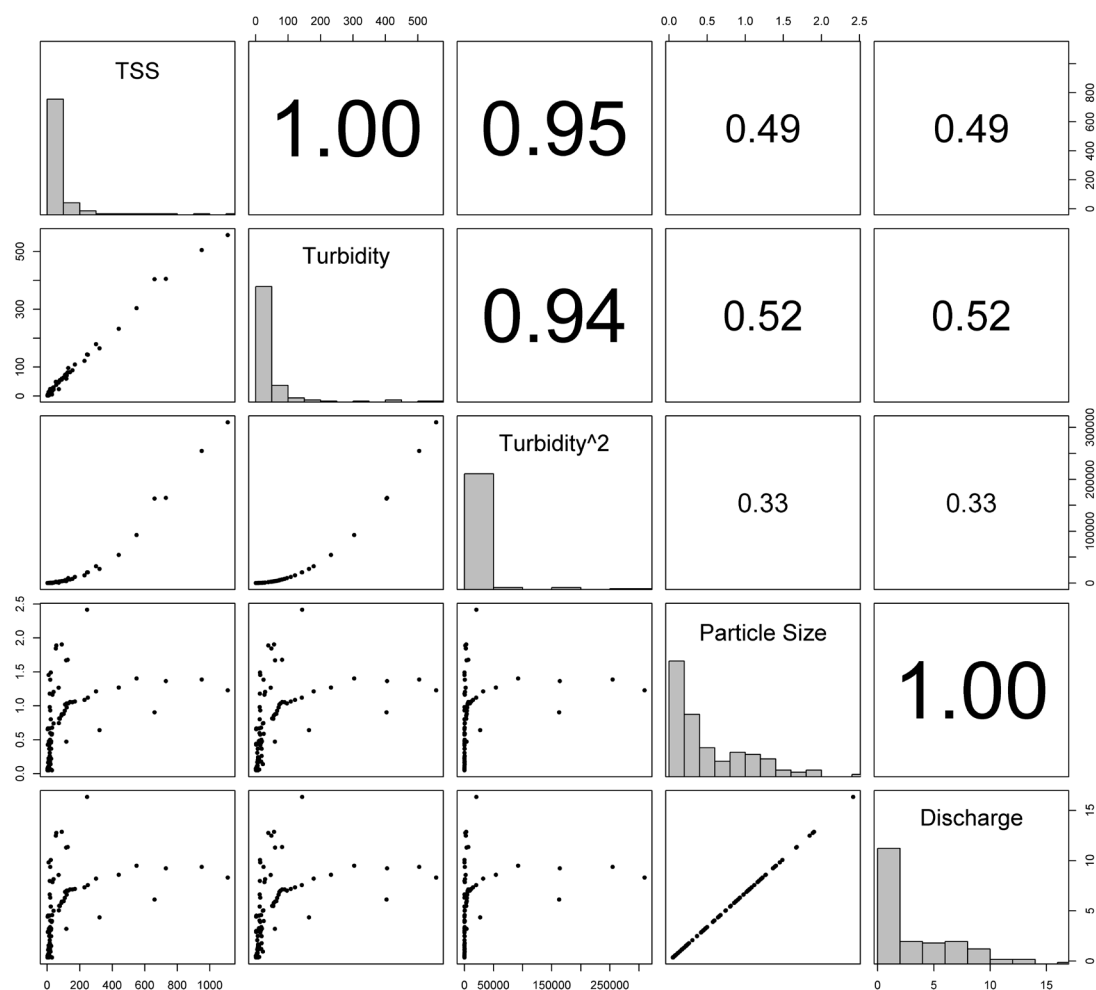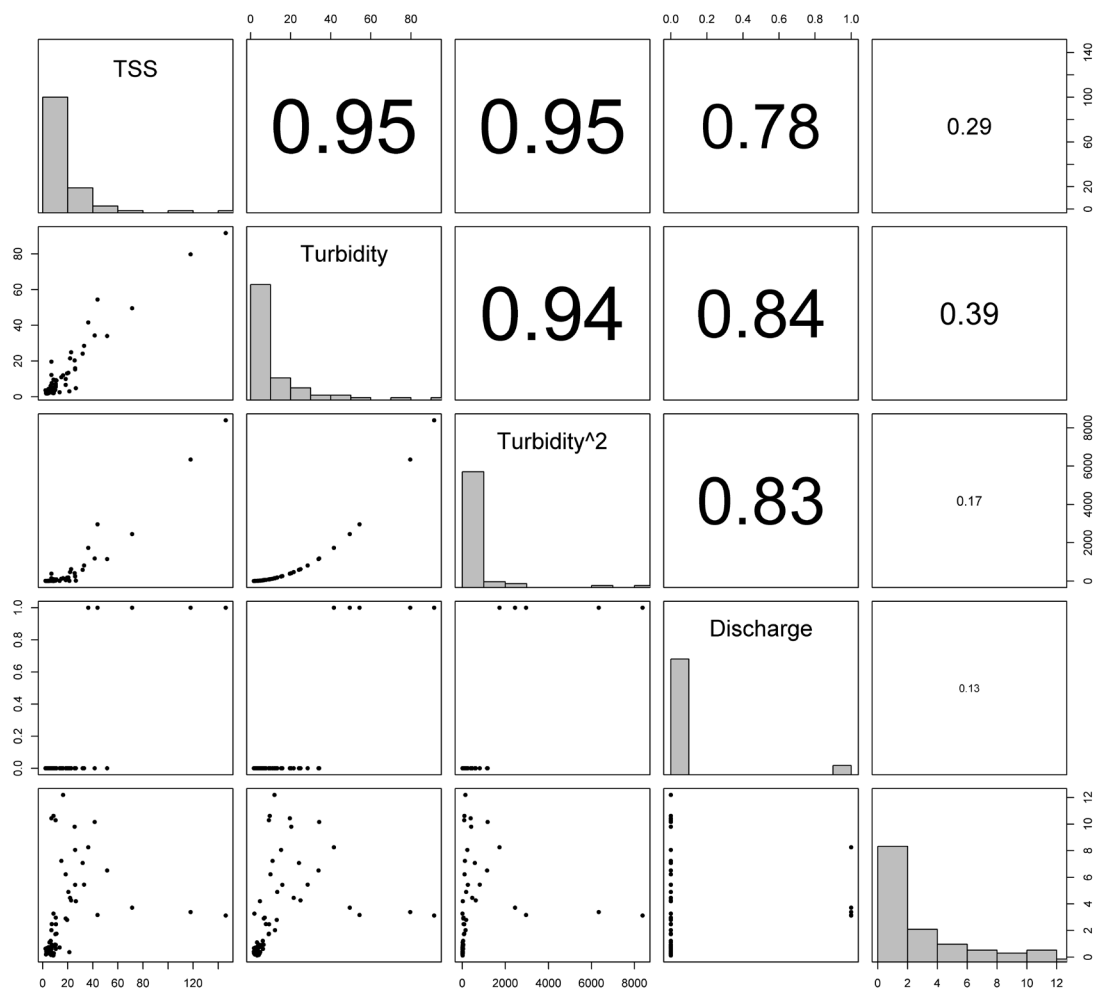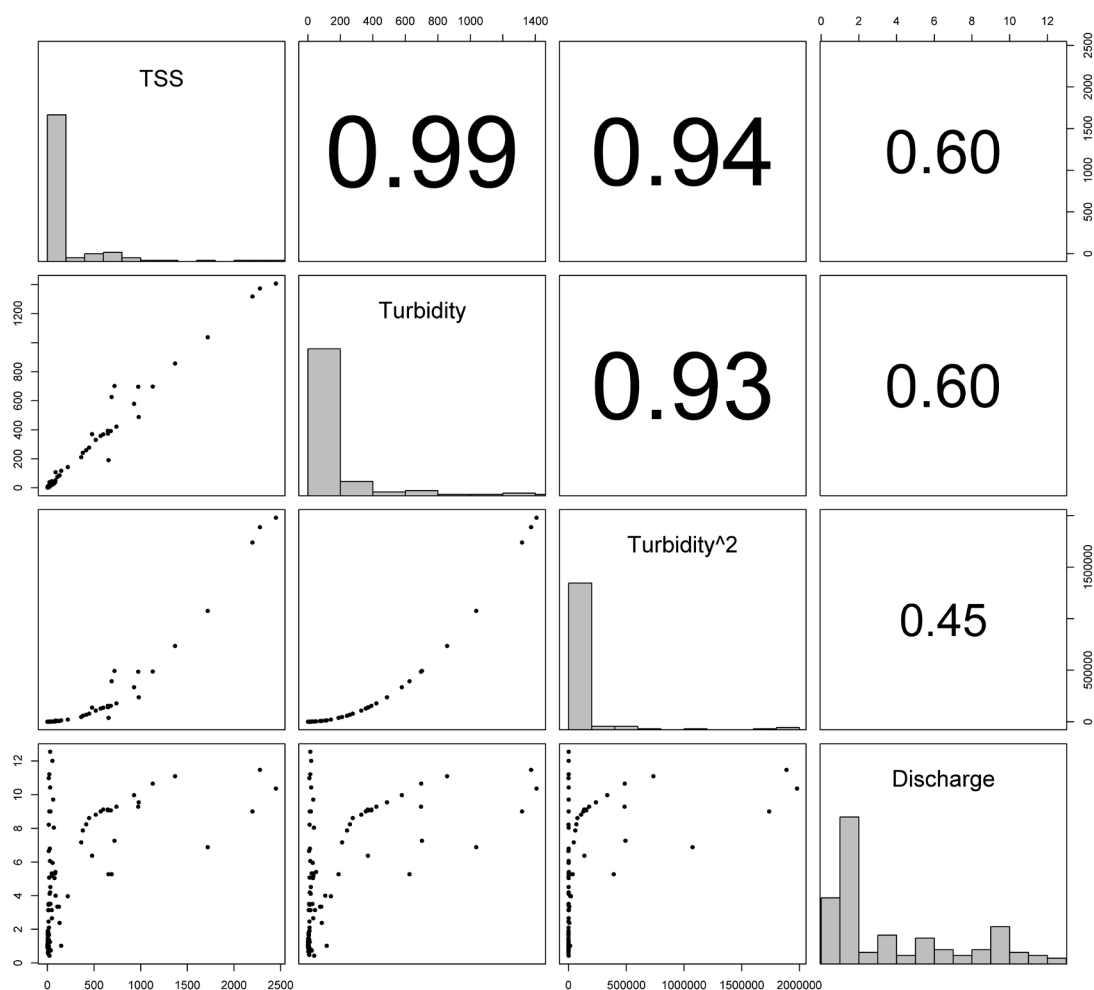
Appendix C. Correlation matrices.
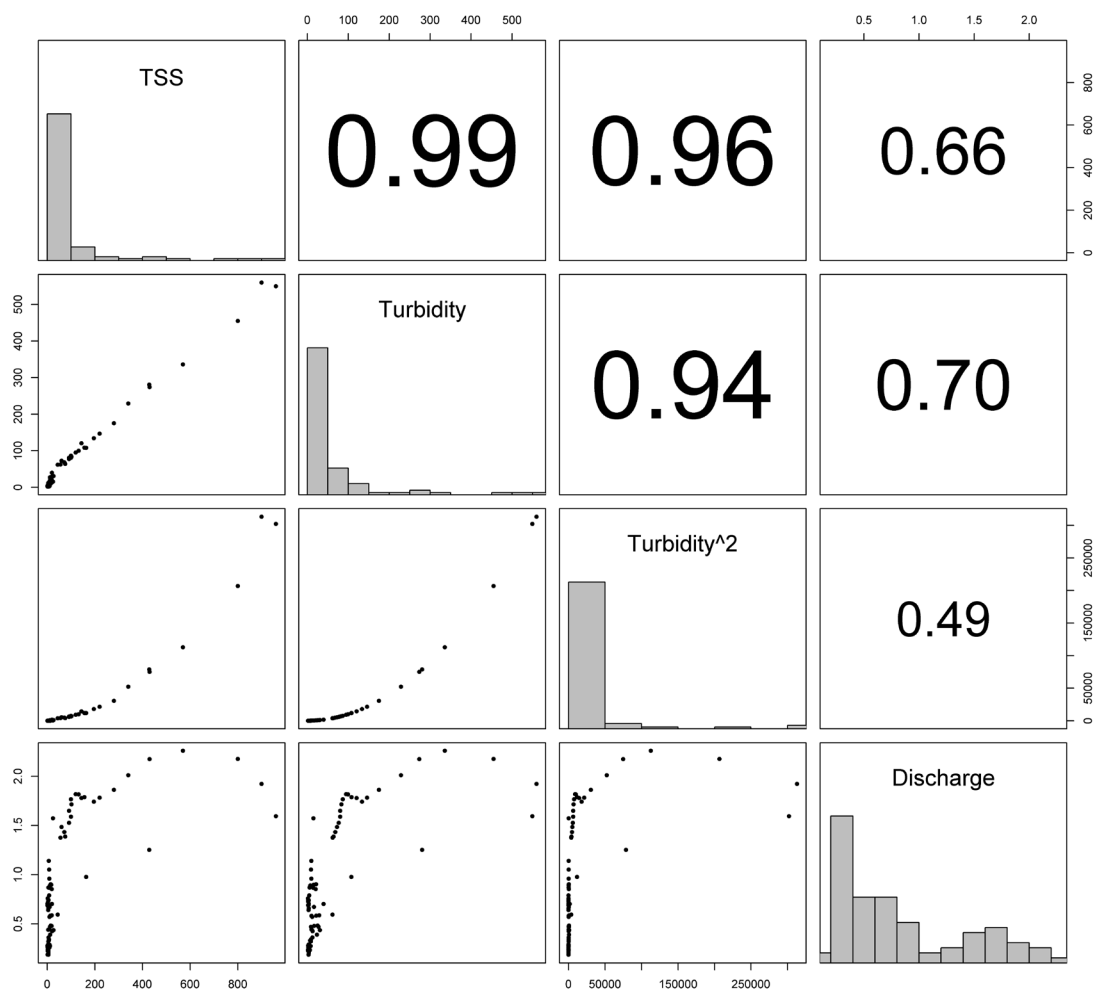
Figure C-1. Correlation matrix for TSS (mg/L), turbidity (NTU), turbidity squared (NTU), categorical variable when turbidity is >= 40 NTU and discharge (m^3/s) at site 1

Figure C3.2.   Correlation matrix for TSS (mg/L), turbidity (NTU), turbidity squared (NTU), categorical variable when turbidity is >= 40 NTU, particle size (1.9-2.25 μm) % of total volume and discharge (m^3/s) at site 2

Figure C-3. Correlation matrix for TSS (mg/L), turbidity (NTU), turbidity squared (NTU), categorical variable when turbidity is >= 40 NTU and discharge (m^3/s) at site 3

Figure C-4. Correlation matrix for TSS (mg/L), turbidity (NTU), turbidity squared (NTU), categorical variable when turbidity is >= 600 NTU and discharge (m^3/s) at site 4
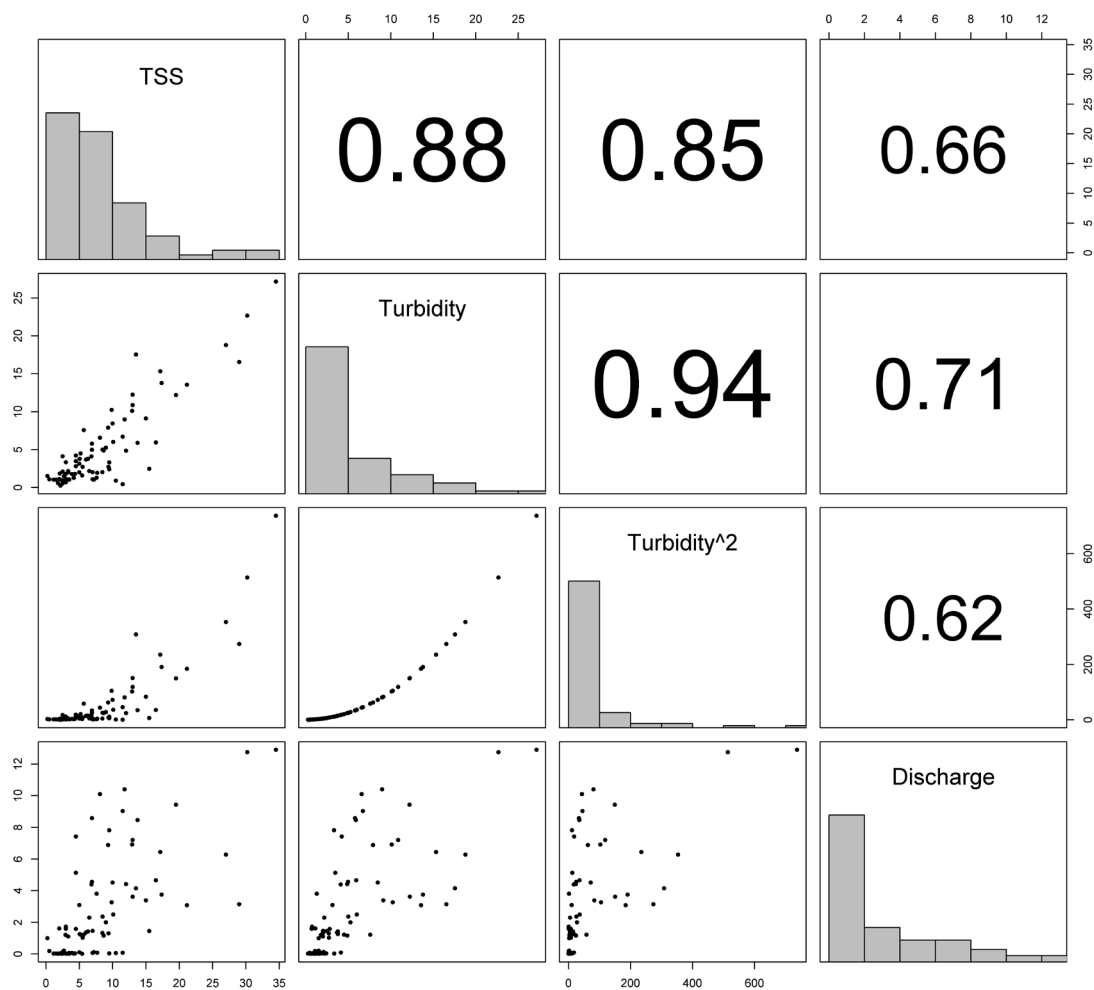
Figure C-5. Correlation matrix for TSS (mg/L), turbidity (NTU), turbidity squared (NTU) and discharge (m^3/s) at site 5
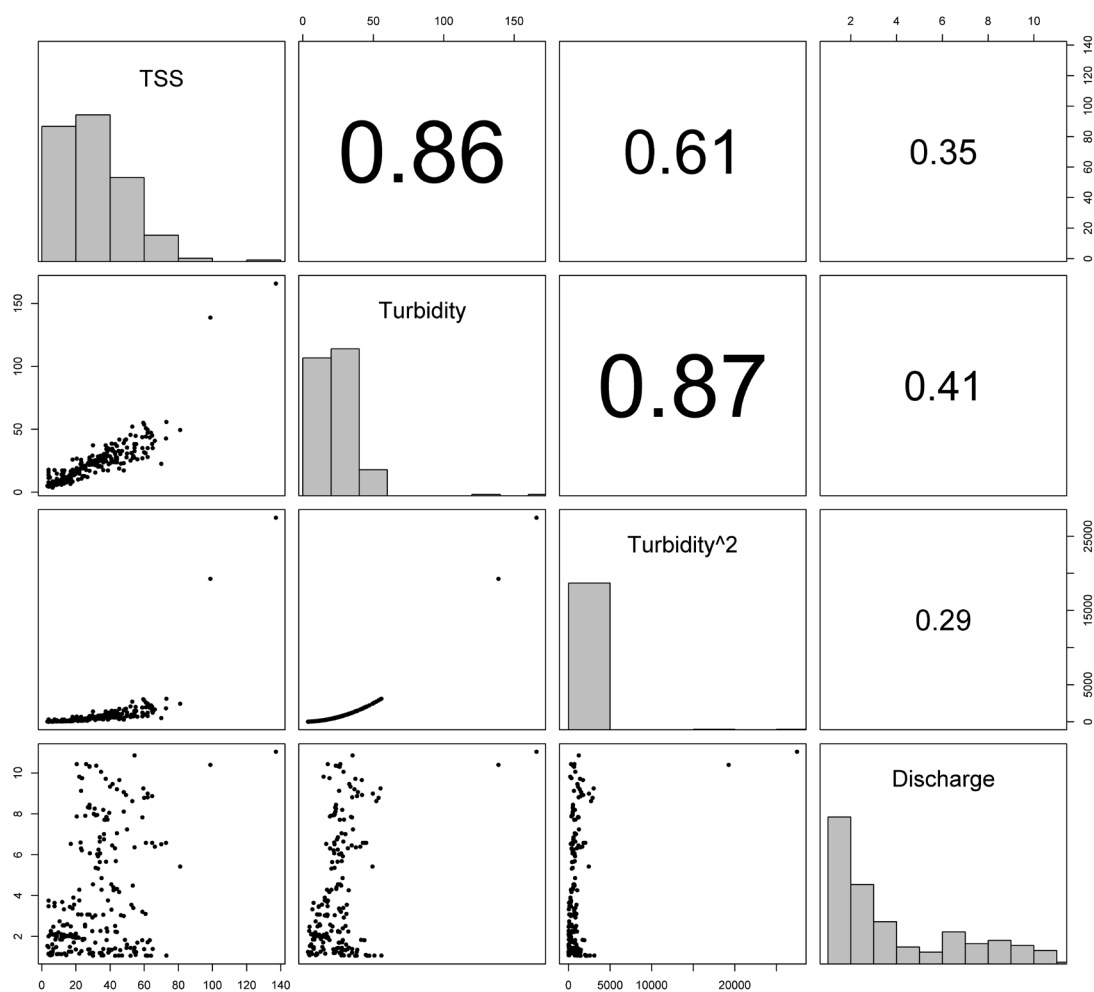
Figure C-6. Correlation matrix for TSS (mg/L), turbidity (NTU), turbidity squared (NTU) and discharge (m^3/s) at site 6